CHAPTER 2
LITERATURE REVIEW

PART I: Waste Stabilisation Ponds

2.1 Introduction

Waste stabilisation ponds (lagoons) are a low-rate form of sewage treatment which can be used either as a partial or a full treatment system. In this way they differ from other low-rate systems such as reed beds and leachfields, which are only suitable for partial, usually secondary or tertiary, treatment. Boller (1997) defined low-rate systems by their low loading rates, oxygenation rates and biomass concentration, comparing them to high-rate systems such as trickling filters and activated sludge systems. Low-rate systems have a lower requirement for energy and skilled operation, but a higher requirement for land.

A waste stabilisation pond system is a set of connecting basins of varying area and depth inside which biological processes break down the wastewater at a natural rate. For ponds with no artificial aeration (unassisted ponds), process control is at the outset by precise design and planning; after that, the performance is subject to the forces of nature such as temperature, wind, sunlight and the biological interactions of micro-organisms.

Although waste stabilisation ponds are simply constructed, their effectiveness depends upon a complex interaction of physical, chemical and biological processes. Ponds within a system have been classified according to the principal biological process, for example: aerobic, anaerobic, facultative, macrophyte, high-rate algal, fish pond etc. Ponds have also been classified by the treatment objective: primary, secondary, tertiary, maturation, or polishing; or even by their hydraulic regime: continuous discharge, intermittent discharge, total containment (Middlebrooks *et al.*, 1977; Water Environment Federation, 1998). Several different ponds are usually used together in series to provide a complete treatment system.

2.1.1 Terminology

Despite 50 years of development in this field, the terminology of ponds continues to be inconsistent. For example, in the United States, all the ponds in a series are called facultative ponds, even if some of them are aerobic (US Environmental Protection Agency, 1983). The terms “maturation”, “tertiary”, “aerobic” and “polishing” are used interchangeably for the same type of pond; (though an “aerobic” pond in the United States is known as a “high-rate algal pond” elsewhere). Waste stabilisation ponds are frequently referred to as “sewage lagoons” which causes much confusion. This situation inevitably contributes to the misinterpretation of data and design standards and the confusion becoming even greater when pond data are presented in different languages.

Table 2.1. Some synonyms highlighting the different use of terminology

anaerobic pond	sedimentation pond
primary facultative pond	facultative first cell oxidation pond primary lagoon
secondary facultative pond	oxidation pond facultative pond
maturation pond	secondary facultative aerobic pond oxidation pond low-rate aerobic pond polishing lagoon tertiary lagoon
high rate algal pond	aerobic pond

2.2 Anaerobic ponds

Anaerobic ponds are used as a primary treatment process. The pond is devoid of oxygen and designed to reduce the wastewater BOD and solids concentrations by sedimentation and anaerobic digestion. They are particularly useful at reducing high concentrations of BOD and SS from agricultural and food processing wastewaters. Anaerobic digestion principally occurs in the sludge at the bottom of the pond, inevitably converting organic load to methane and carbon dioxide and releasing some soluble by-products into the water column (eg. organic acids, ammonia). It is thought that the waste gases carry anaerobic bacteria into the water column, thus perpetuating anaerobic digestion throughout the pond.

At temperatures below 15^{o}C, the digestion processes slow down and the dominant process is thought to be sedimentation (Pearson, 1990). Anaerobic ponds are usually more than 2 m deep for sludge storage capacity. The hydraulic retention time depends on the volumetric BOD loading required (g/m^3.d) for the climate and can be up to 20 days (WEF, 1998). Anaerobic ponds are very beneficial to the overall system as they can reduce much of the organic load (between 40-70% BOD) using less than half the area of other types of pond (Mara and Pearson, 1998). They also reduce the problems associated with sludge accumulation and solids feedback in a following facultative pond. The high efficiency of BOD removal combined with the partial mineralisation of organics experienced in an anaerobic pond allows for smaller subsequent ponds thereby reducing the overall land requirements (Mara and Mills, 1994). The main disadvantages of anaerobic ponds are the risk of odour and the increase in ammonia and sulphide concentrations caused by the anaerobic processes (Pearson *et al.*, 1987b).

2.3 Maturation (Aerobic) ponds

Maturation ponds are primarily designed for tertiary treatment: the removal of pathogens, nutrients and possibly algae. They are very shallow (usually around 1 m depth) to allow light penetration to the bottom and aerobic conditions throughout the whole depth. The ponds follow a secondary treatment eg. a facultative pond. The size and number of maturation ponds needed in series is determined by the required retention time to achieve a specified effluent pathogen concentration. In the absence of effluent limits for pathogens, maturation ponds act as a buffer for facultative pond failure and are useful for nutrient removal (Pano and Middlebrooks, 1982; Mara and Pearson, 1998).

2.4. Facultative ponds

2.4.1 Introduction

Facultative ponds perform both primary and secondary treatment. Primary facultative ponds receive raw wastewater, usually after screening, and therefore store wastewater solids as sludge. Secondary facultative ponds follow primary treatment e.g. an anaerobic pond, so do not receive the same settleable solid load.

Facultative ponds are characterised by a permanent anaerobic layer at the bottom where the sludge accumulates, and an aerobic layer on the surface oxygenated by the photosynthetic action of algae and wind aeration. Depending on the oxygen uptake rate of the system, the aerobic layer may become anoxic at night when photosynthesis stops.

2.4.2 Physical design

Facultative ponds are typically shallow (1-2 m deep) to allow adequate light penetration and deep enough to prevent the emergence of macrophytes. Primary facultative ponds may have a deeper area around the inlet to provide extra sludge storage capacity.

Primary facultative ponds are usually rectangular, with a length to breadth ratio of 2-3:1 to prevent excessive sludge build-up around the inlet. Higher ratios (up to 10:1) reduce short-circuiting and may be suitable for secondary facultative ponds, where sludge accumulation around the inlet is less of a problem (Mara and Pearson, 1998). Facultative ponds should be regular in shape with no isolated areas where circulation might be impeded and weed growth encouraged. If the area required is very large, then multiple cells in parallel are used to reduce short-circuiting, reduce bank erosion from wave action and to allow more flexible operation; multiple cell units however, incur increased construction costs and land requirements. Baffles have also been used on facultative ponds to reduce short-circuiting (Pearson, 1996; Muttamara and Puetpaiboon, 1997).

2.4.3 Facultative pond process biology

As flow enters the pond, a portion of the load enters the liquid and the rest settles to the bottom as sludge. In the sludge layer the degradation is performed by anaerobic bacteria; at the surface, the waste is degraded aerobically by heterotrophic bacteria (if sufficient dissolved oxygen is present). The latter process relies on the mutualism between the algae and the heterotrophic bacteria. The algae, via photosynthetic processes, are the principal source of the dissolved oxygen which is used by the bacteria to oxidise the waste. The algae utilise bacterial by-products such as phosphate, carbon dioxide and inorganic nitrogen to make cellular organic materials. The community in primary facultative ponds mainly consists of micro-algae, facultative bacteria and anaerobic bacteria; though a wide variety of other organisms such as protozoa, rotifers, fly larvae, viruses and fungi may also be present depending on the time of year (Gloyna, 1976). A diagrammatic representation of the major interactions in facultative ponds as given by Metcalf and Eddy (1991) is reproduced as Figure 2.1.

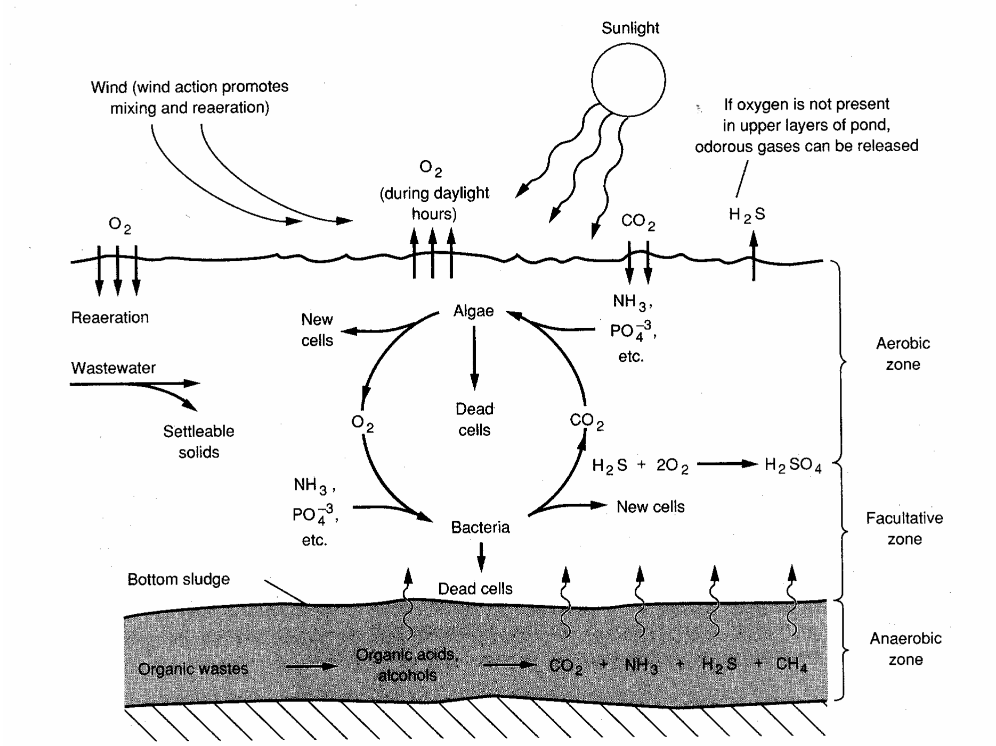

Figure 2.1 Diagrammatic Representation of Processes in a Facultative Pond (from Metcalf & Eddy(1991))

2.4.3.1 Algae

The algae occupy the surface layer where light can penetrate (the photic zone) and give facultative ponds their characteristic green colour. The aerobic zone is typically 40 cm deep (Mara and Pearson, 1986), but may become deeper in response to lower organic loading. All the algae found in facultative ponds are organic pollution tolerant (Palmer, 1969), but motile genera such as *Chlamydomonas, Euglena,* and *Phacus* tend to dominate (Mara and Pearson, 1998). This is due to their ability to adjust their vertical position to optimise light access in the turbid pond water. The motile algal population tends to form a narrow band about 20 cm thick which moves up and down the water column during the day and disperses at night (Mara and Pearson, 1986). Non-motile algal genera such as

Chlorella and *Scenedesmus* also are abundant, but only dominate at lower organic loading. Competition for light is the main variable affecting the speciation of algae in facultative ponds. Pearson *et al.* (1987b) showed that these non-motile algae will out-compete the motile genera if they can take advantage of the sunlight.

In addition to providing dissolved oxygen and taking up nutrients, some algae have been shown to effect treatment by assimilating organic matter (Pearson *et al.*, 1987b). For example, species of *Euglena* and *Chlamdymonas* have a flexible metabolism with the option of heterotrophic assimilation in both light and dark conditions (Pearson, 1990).

The algal biomass, as measured by the chlorophyll-a concentration, in a thriving facultative pond can range between 500–2000 µg/l; a minimum of 300 µg/l has been suggested for a correctly operating pond (Mara and Pearson, 1986).

2.4.3.2 Bacteria

The heterotrophic bacteria found in the aerobic zone are typically the same type as those found in other aerobic forms of sewage treatment. The most frequently isolated bacteria include *Beggiatoa alba, Sphaerotilus natans, Achromobacter, Flavobacterium, Pseudomonas* and *Zoogloea* spp. (US Environmental Protection Agency, 1983). In temperate climates, mesophilic[1] and psychrophilic[2] bacteria dominate in summer and psychrophilic and psychrotrophic[3] bacteria dominate in winter (Townshend and Knoll, 1987).

Below the algal zone, where dissolved oxygen is low or zero, and only long wavelength light (750-900 nm) penetrates (chlorophyll-a absorbs at 663 nm), anaerobic photosynthetic bacteria such as the *Chromatiaceae* (purple sulphur bacteria; retain S as granules) and the *Chlorobiaceae* (green sulphur bacteria; release S) proliferate. These

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[1] thrive in temperatures between 15 –35 °C
[2] grow in temperatures between 0–15°C
[3] grow below 0°C

sulphur bacteria use H_2S instead of water as part of photosynthesis, producing elemental sulphur rather than oxygen as a by-product. In the presence of light they can also oxidise sulphide to sulphate (Pearson, 1990). These bacteria play an important role in controlling odour releases from facultative ponds. Photosynthetic non-sulphur bacteria also grow in this region of the pond, for example the *Rhodospirillaceae* (purple) and the *Chloroflexaceae* (green), using anoxygenic photosynthesis to oxidise organic substrates. The purple bacteria contain bacteriochlorophylls a and b and can be pink, purple or red in colour; consequently this zone usually has a characteristic pink/purple colour. A purple colouration on the surface of a facultative pond indicates anaerobic conditions in the upper layer.

At the bottom of the pond is the sludge layer which is permanently anaerobic. Here, a wide range of proteolytic bacteria such as *Streptococcus*, *Clostridium*, and *Staphylococcus* can degrade proteins and amino acids to simple organic acids, ammonia, carbon dioxide, hydrogen and hydrogen sulphide. Acetogenic bacteria are obligate anaerobes which use the hydrogen and carbon dioxide to produce acetate. Methanogenic bacteria such as *Methanobacterium* and *Methanospirillum* convert acetate and hydrogen to methane. Methanogenesis can remove significant quantities of BOD from the sludge, but requires a narrow pH range (6.4-7.2) and moderate to high temperatures (above 15^oC) (Mara and Pearson, 1986). Sulphate-reducing bacteria (eg. *Desulfovibrio*) also inhabit the sediments, converting sulphate to sulphide. This sulphide can then be oxidised to sulphur or sulphate, either by the purple bacteria in the facultative zone (Mara and Pearson, 1986), by aerobic bacteria such as *Thiobacillus* in the surface layer, or chemically by direct combination with dissolved oxygen.

2.4.4 Aerobic and anaerobic conditions

In the liquid column of a facultative pond aerobic and anaerobic conditions may alternate. The rate at which the pond water oscillates between aerobic and anaerobic conditions principally depends on the balance between the respiratory and photosynthetic rates, which in turn depend on temperature, availability of light, relative number of algae and bacteria present and the availability of nutrients. Anaerobic conditions tend to occur: during the night when the algae are respiring; during calm hot weather when the solubility of oxygen in water is low and there is insufficient wind mixing; or all the time if the pond is overloaded. Aerobic conditions usually resume when: the morning sun initiates photosynthesis; when windy conditions aerate the surface or mix stratified layers; or when the loading is reduced to suit the climatic conditions.

As algae are the main source of dissolved oxygen, the concentration reaches a peak when the intensity of incident sunlight is at its greatest. In the presence of an abundant algal population the relative rate of photosynthesis can commonly result in supersaturated dissolved oxygen concentrations during the day. In calm conditions the formation of the algal band shades out the lower areas and the non-motile algae sink out of the photic zone and start to respire. Without adequate mixing anaerobic conditions can occur directly underneath the band. In unassisted ponds only wind-mixing and thermal overturn can destroy the stratification enabling the non-motile algae to re-enter the photic zone. In cold and temperate climates the algae may not develop all year and thus the oxygen supply may be unreliable (Cauchie *et al*., 2000).

Mixing and temperature were argued as the most important physical factors for the reoxygenation of facultative ponds by Marais (1970). Occurring principally by wind action and thermal mixing at night, the intensity of mixing affects the kinetic rate of bacterial degradation in the pond liquid, and the growth, type and concentration of algae. The temperature affects not only the biological rates, but also the physical mixing ability of the wind. Marais (1970) reported that at Matero Pond in Zambia the wind did not appear to be as effective for mixing during the warmer months as in the colder months. According to Fair and Geyer (1954) as reported by Marais (1970), as average water temperature increases, the wind energy required to destroy a 1degC difference in temperature between two layers of water increases significantly. Direct rainfall has also been suggested as a source of mixing and oxygenation: Ellis (1983) suggested that rainfall falling at 5 mm / h for 2 hours can add 30 kg dissolved O_2 / ha.

2.4.5 Pond failure

Pond failure occurs when the reoxygenation processes are insufficient to generate aerobic conditions during daylight hours. Failure is indicated by changes in biology: the first stages are accompanied by a reduction in algal diversity and eventually only motile algae such as *Chlamdymonas* and *Euglena* may remain (Almasi and Pescod, 1996). When the algae disappear completely, anoxic or anaerobic conditions prevail and the purple photosynthetic bacteria become visible at the surface (Li *et al.*, 1991; Lai and Lam, 1997). The pond may become fully anaerobic, indicated by a black colouration, elemental sulphur deposits (from the oxidation of hydrogen sulphide by anaerobic photosynthetic bacteria), and possibly hydrogen sulphide odour from anaerobic sulphate reduction.

The loss of the algae may be due to a number of factors, but is mainly due to the overloading of solids, either from the wastewater or the sludge, which block out light (BOD in itself does not retard algal growth) (Parker and Skerry, 1968). The algae may also be destroyed at low or moderate loadings by other factors such as grazing, ammonia and sulphide toxicity (Shillinglaw and Pieterse, 1977; Konig *et al.*, 1987); or algal parasites (Lawty *et al.*, 1996). Climate also has a very important part to play.

Solids blocking out the sunlight seriously affect the algae; the sources of these solids include excess colloidal solids in influent wastewater; solids feeding back from the sludge; and other algae (self-shading). Increasing the organic load can stimulate gas production in the sludge which causes the eruption of solids and ammonia into the upper layers (Parker and Skerry, 1968). At lower loadings *Daphnia*, rotifers and large protozoan blooms can consume the algal population within days (Cauchie *et al.*, 2000).

Ammonia and sulphide, which are algal poisons at high concentrations, are present in the raw wastewater and generated by in-pond biological processes. Toxicity is pH and temperature dependent: sulphide toxicity increasing with decreasing pH and increasing temperature; while ammonia toxicity increases with pH and temperature (Pearson *et al.*, 1987b). Ammonia toxicity is greatest in the surface layers, where the pH is higher (Pearson, 1990). The toxicity is species dependent: eg. *Chlorella* is four times more tolerant to ammonia (356 mg NH_4-N at pH 8.5) than *Chlamdymonas* and *Euglena* (Pearson, 1990). This may cause problems if very high ammonia wastewaters are applied to facultative ponds, as the dominant motile algae are more sensitive to ammonia. *Chlamdymonas* is the least sensitive genus to sulphide toxicity, tolerating four times the concentration of *Euglena* (Pearson *et al.*, 1987b). Consequently, *Chlamdymonas* is usually the last algal genus to survive in anaerobic conditions (Almasi and Pescod, 1996).

Season has a strong influence on algal populations in temperate climates; the lower temperatures and shorter daylength experienced in winter may be insufficient to support an algal population at all (Cauchie *et al.*, 2000). At moderate to high loadings nutrients are usually provided in excess from the wastewater (domestic wastewater usually has high concentrations of nitrogen, phosphorus and carbon compounds), so the algae are usually light-limited. Racault (1993) studied three ponds in SW France which failed during the winter. The three ponds failed at the same time after the death of the algae in late autumn; insufficient surface aeration during the calm winter led to odour problems.

All the factors mentioned affect the organic load which may be applied to the system before failure, but the critical combination of these factors depends on the pond location and environment. The outcome of pond failure includes: reduction in BOD and SS removal efficiency due to sludge solids entering the effluent; objectionable odours due to anoxic conditions; the production of sulphide and sulphur. However, Almasi and Pescod (1996) showed that soluble BOD removal was unaffected by the change in ecology because the pond bacteria can consume large quantities of organic matter in both anoxic and anaerobic conditions.

Data collected in Brazil and Portugal suggests that a consistent mean water column chlorophyll concentration below 300µg/l indicates an unstable system which may be heading towards failure (Pearson, 1996).

2.4.6 Sludge accumulation

Primary facultative pond sludge originates from both the wastewater and the internal production of algal and bacterial cells. In temperate and cold climates the internal production mechanisms contribute most significantly during the summer (Schneiter *et al.*, 1983; Iwema *et al.*, 1987). The rate of accumulation of sludge depends on: the wastewater characteristics (in particular, the concentration of settleable solids in the wastewater); the ambient environment (eg. temperature and sunlight); operational variables such as inlet and outlet configuration, depth and geometry of the pond; and the organic loading applied (Schneiter *et al.*, 1983; Carre *et al.*, 1990). The volume of the accumulated sludge is continuously reduced by compression and anaerobic degradation processes, and the net rate of accumulation reduces significantly over time (Nelson, 2002).

The rate of anaerobic digestion increases approximately sevenfold with each 5degC rise in temperature (Marais, 1970). In hot climates a steady-state equilibrium can establish where the rates of accumulation and degradation balance, so sludge removal can be avoided completely. However, in cold and temperate climates the degradation rate is slower: sludge tending to accumulate in winter and digest only in summer (Parker and Skerry, 1968; Schneiter *et al.*, 1983). In these climates the steady state is never established, so additional storage depth must be provided and desludging will be required at some point. Mara (1976) suggested that the desludging interval is 10 years or more. Carre *et al.* (1990) reported the average desludging interval for 12 French ponds was between 6-7 years when the sludge occupied 30% of the pond volume (three of the ponds did not require desludging after more than ten years). The average accumulation of sludge in the French ponds was 2.8 cm per year (or 0.13 m^3/person.yr). The Mangere ponds in Auckland NZ, as reported by Lawty *et al.* (1996) accumulated an average of only 30 cm over 34 years of operation, while Canadian and Alaskan pond accumulation, as reported by Schneiter *et al.* (1983), was between 0.073-0.146 m^3/person.yr

In the sludge layer the anaerobic degradation processes release gas: Parker and Skerry (1968) found the gas to be 36-87% methane, 0-19% CO_2 and differing amounts of others including N_2. Bryant and Rich (1984) found the composition of the gas to be: 75% methane, 12% CO_2 and 7% N_2. Marais (1970) stated that sludge methane production can allow up to 30% reduction of the influent BOD. The ammonia and organic acids produced are transported from the interstitial fluid to the overlying water by diffusion or advection induced by compaction. These feedback processes vary significantly with temperature, sludge type and volume (Bryant and Bauer, 1987). Methane production was found to be ten times greater in summer than winter in temperate climates by Iwema *et al.* (1987). Increasing organic load and temperature stimulates gas production which creates motion in the sludge which becomes less dense and may erupt solids into the upper layers of the pond (Parker and Skerry, 1968; Iwema *et al.*, 1987).

There are complex interactions between the sludge and the water above (Parker and Skerry, 1968). Adsorption and remineralisation processes occur at the interface and depend on conditions such as pH, redox and temperature (Nameche *et al.*, 1997). Over time, the lower layers of the sludge get compacted and dense, and the sediment-water exchanges slow down. The % volatile solids, particulate TKN, and temperature decrease in the deeper sludge layers; while the % dry solids and ammonia concentration increase as a result of compaction and remineralisation processes (Nelson and Jimenez, 2000). After 10 years, the dry solid content may increase by up to 10-fold (Carre and Baron, 1987). At low temperatures storage is the dominant process, at medium temperatures feedback is dominant process, and at high temperatures methane production is the dominant process (Bryant and Rich, 1984). Heavy metals may accumulate and concentrate during digestion: Nameche *et al.* (1997) found these were 1000-7000 times more concentrated in the sludge than in the overlying water, while the nitrate concentration in the sediment was always lower than in the overlying water (evidence of denitrification).

Sludge accumulates unevenly across the pond, the thickest layers usually occurring at the inlet, outlet, near the embankments and in least mixed areas (Schneiter *et al.*, 1983; Carre *et al.*, 1990; Nameche *et al.*, 1997). Channels may form as a result of the inlet configuration and the scouring effects of the influent flow (Carre *et al.*, 1990; Saqqar and Pescod, 1995). Accumulated sludge peaks may also "slide" thereby mixing and levelling themselves out (Nelson and Jimenez, 2000).

The sludge composition varies across the pond: inlet sludges, consisting of large particles of mineral clay and silt, are higher in % dry solids (Carre and Baron, 1987), while outlet sludges, which include high quantities of algae, consist of fine organic particles. Ammonia concentration is twice as high in inlet sludge (wastewater solids are richer in nitrogen than algal solids) (Nelson and Jimenez, 2000), although the %VS in both is about the same (40-60%) (Parker and Skerry, 1968).

Characteristics of primary facultative pond sludges were reported by Schneiter *et al.* (1984) as 1.2 – 15 % dry solid; %VS 29 – 93 %; FC 2.4 – 4.9 x 10^3 / 100 ml; these values are within the range reported for untreated primary sludge (though the range is wide). Pond sludges are similar to other sludges except the VS/TS ratio is generally higher and the FC are significantly lower (Reed *et al.*, 1988). Pond sludge is typically high in VS, and low in dry solids, FCs, nitrogen and phosphorus, and it is poorly stabilised even after long residence on the pond bottom. The unstable quality is thought to be as a result of the continuous supply of fresh solids (Nameche *et al.*, 1997). FC in the sludge decrease over time by 3.3-4 log over 30 cm depth (Carre and Baron, 1987; Nelson and Jimenez, 2000). Nameche *et al.* (1997) found that pond sediments had fewer FC than manure and other sludges spread on agricultural land in France (a typical FC number in wastewater sludges is 10^7 / 100 ml (Reed *et al.*, 1988)). Algal sludge is 6-8% nitrogen by weight (Golueke *et al.*, 1957).

Sludge accumulation may cause problems for the operation of the facultative pond. The reduction of effective volume (hence hydraulic retention time) affects performance (Nameche *et al.*, 1997); also feedback to the water column augments the nutrient and organic load. Sludge can also be a source of odour and resuspended solids entering the water column and effluent. Sediment oxygen demand from the sludge may be significant, both from biological respiration and chemical oxidation processes on the surface of the sludge: Nameche *et al.* (1997) calculated that 1-3 g $O_2/m^2.d$ at 20 ^oC was demanded from the sludge layer. Sludge activity was shown by Parker and Skerry (1968) to have no significant detrimental effect on algae provided the solids were not mixed into the liquid.

Sludge removal is the most demanding operation for otherwise low maintenance waste stabilisation ponds (Carre *et al.*, 1990); but sludge disposal charges represented 25% of the total operating cost of all wastewater treatment in UK in 1995 (Hosetti and Frost, 1995). The *average* accumulation per year is not likely to be the best predictor of the desludging interval because most of the sludge accumulates near the inlet. The use of "m^3/ person.yr." units has also been argued against because it does not take into account wastewater characteristics (Saqqar and Pescod, 1995).

Desludging using a raft-mounted pump is recommended by Carre *et al.* (1990): draining the pond is not required and the collected sludge is quite wet and easy to pump. In French ponds, pumping out of sludge takes between 0.5-15 days (average 3 days) depending on the size of the pond.

2.4.7 BOD removal

The ability of a facultative pond to remove BOD is the primary factor by which its operation is assessed. Mara & Pearson (1998) and the US EPA (1983) report BOD removal to range between 50-90%. This wide variation reflects the variety in design and a heavy reliance on local climatic conditions.

It has been commonly noted that the BOD effluent concentration does not vary seasonally in temperate climates as would be expected (Marais, 1970; Mara, 1976; Racault *et al.*, 1995): i.e. the pond BOD would be expected to be higher during winter because the rate of degradation is low. However, during the winter little degradation takes place in the sludge and so there is low feedback to the pond water. During the summer, the digestion of the sludge accumulated in the winter feeds back significant quantities of BOD, which offsets the high degradation rate in the pond. This effectively evens out the effluent BOD concentration. However, the effluent BOD is augmented by algal solids in the summer when between 70-90% of the effluent BOD may be due to algae (Mara and Pearson, 1998). German researchers, as reported by Bucksteeg (1987), determined the BOD contribution of algae in units of chlorophyll-a as follows: 100 μg chlorophyll-a $\equiv$ 3 mg BOD_5.

2.4.8 Suspended solids removal

Primary ponds in temperate climates may remove between 60-70% of influent SS by sedimentation processes (Schneiter *et al.*, 1983)[4]. However, SS in facultative pond effluents can be very high (>100 mg/l) due to algal cells; these high concentrations may be only experienced in summer in temperate climates (Reed *et al.*, 1988).

SS removal is affected by: the settleable solids concentration in the influent; the accumulation of sludge; solids feedback into the water column; and the washing out, rather than sedimentation, of algal biomass. Particularly in temperate climates, the spring and autumn overturn may also cause resuspension of sludge solids which enter the effluent (US Environmental Protection Agency, 1983). Most design equations are not explicit for suspended solids removal.

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[4] based on the primary cells of the Logan and Corinne ponds in Utah, USA

2.4.8.1 Algal suspended solids

By their nature, facultative ponds produce large quantities of algae. In normal operation, over 90% of solids leaving the facultative pond during summer will be due to algae. Mara *et al.* (1992) reported that the algal concentration in facultative pond effluents range between 1000-1500 μg chl-a / l in hot climates and may reach this in temperate zones during the summer where the algal contribution to the suspended solids concentration can reach 40-100 mg/l. During the growing season therefore, removal efficiency for suspended solids in facultative ponds is meaningless. The movement of the algal band can lead to diurnal fluctuations in effluent quality depending on the take-off height of the outlet. Column samples taken at the effluent point can give a good estimation of diurnal effluent quality. To reduce the algal concentration leaving the facultative pond Pearson (1990) suggests positioning the outlet at a depth of 40-50cm, thus avoiding the algal band.

Some authors consider that "algal SS and BOD" should not be subject to conventional effluent requirements (eg. USEPA, 1983; Mara, 1996). In some instances standards for BOD and SS concentrations have been relaxed for effluents from pond systems (eg. in the United States, France, Germany, European Union). This is because algal cells are not readily biodegradable (Golueke *et al.*, 1957; Fitzgerald, 1964), have low settling velocities so do not readily settle in streams and may be dispersed over a wide area before they exert an oxygen demand on the watercourse. In addition, continuing photosynthesis may result in net oxygen input into a receiving watercourse and the algae may promote an increase in the productivity of the aquatic system. At high concentrations however, uncontrolled discharge of algae is undesirable as they can deplete oxygen reserves at night. The Environment Agency for England and Wales do not permit relaxation of their standards for waste stabilisation ponds: their reasoning based on consideration of eutrophication, turbidity, night-time respiration and deleterious oxygen demand exerted in areas of slow moving water where the algal cells congregate after morbidity (Morris, 1999).

The German relaxation for waste stabilisation pond discharge incorporates chlorophyll-a as shown in equation 2.1.

$$BOD_R = BOD_E - 0.03\ (\text{Chl-a}) \qquad (2.1)$$

Where BOD_R = required BOD (mg/l)
BOD_E = actual unfiltered BOD (mg/l)
Chl-a = chlorophyll-a concentration (µg/l)

The European Union relaxation as given in the EC UWWTD is: BOD and COD analyses on the effluent may be carried out on filtered samples and SS must not exceed 150 mg/l (Council of the European Communities, 1991). The Scottish Environment Protection Agency (SEPA) has allowed relaxation of the standards in accordance with the EU[5].

2.4.8.2 Removal of algae from effluents

Many techniques have been developed to remove the algae from effluents, these include rock filtration, grass plots, floating macrophytes and herbivorous fish. Also, the use of maturation ponds can reduce the algal concentration considerably provided the system is not overloaded. Maturation ponds generally have lower nutrient concentrations than the facultative pond which they follow and may also have larger communities of algal predators such as *Daphnia*. Mara *et al.* (1992) summarised the typical reduction in chlorophyll-a in maturation ponds in hot climates: reducing to 50% by the end of the first maturation pond and to 10-30 % after the second.

Floating plants have been tried on maturation ponds to shade out the light and cause the algae to die and settle out to the bottom; nutrients released by algal decomposition are taken up by the roots of the floating plants. Water hyacinth and water lettuce have been used in hot climates; duckweed might be useful in colder climates, though all these plants need regular harvesting. Fish are an option for final maturation ponds.

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[5] For the Ponds at Errol, near Dundee (see Section 2.20)

The algal laden effluent may be useful if discharged to soil or grass plots: the algae act as a slow release fertiliser supplying plant nutrients in the right proportions and reducing the risk of groundwater pollution (Mara *et al*., 1992). Submerged porous rock filters where the algae settle out and decompose are used in the United States, as are more ‘high tech’ methods such as: coagulation-clarification, autoflotation, direct filtration and microstrainers. A detailed review of these techniques can be found in US EPA (1983).

2.4.9 Ammonia removal

The concentration of ammonia in domestic wastewater is typically between 9-30 mg/l, but may be higher than 50 mg/l (Konig *et al.*, 1987; Reed *et al.*, 1988). Typically, 60-75% of total nitrogen in wastewater is in the form of ammonia, whilst the rest is organic nitrogen (Reed, 1985; Lai and Lam, 1997). Microbial hydrolysis of organic nitrogen, either in a preceding anaerobic pond or in the facultative pond sediment may increase the ammonia concentration further.

The following pathways have been suggested for ammonia removal in facultative ponds:

1. Ammonia volatilisation (loss as a gas to the atmosphere),
2. Ammonia assimilation into the algal biomass (conversion to organic nitrogen),
3. Aerobic nitrification (conversion to nitrate), and
4. Physical adsorption and sedimentation.

It is currently thought that the first two pathways are the major ones and the other two are minor. There is disagreement, however, regarding the relative importance of the volatilisation and algal uptake pathways.

2.4.9.1 Ammonia volatilisation

Ammonia in water exists in equilibrium between the ionised (NH_4^+) and the free (NH_3) forms. The free NH_3 form is volatile and may be lost to the atmosphere as a gas in a physical process. The equilibrium between the two forms depends on temperature and pH. At pHs above 6.6, NH_4^+ starts to convert to the volatile NH_3 form; at pH 9.2 the two forms are equal in concentration, and at pH 12 all the ammonia is in the NH_3 form (Reed, 1985). A 10degC rise in temperature will also double the concentration of NH_3 at a given pH (Konig *et al.*, 1987) and, like other gases, ammonia is more soluble in water at low temperatures (eg. 6.84 mg/l at 10^{o}C and 5.29 mg/l at 20^{o}C). Consequently, loss by volatilisation is more likely at higher temperatures and higher pH. High pH conditions (>8.0) are common in facultative ponds during periods of intense algal photosynthesis due to the rapid removal of dissolved CO_2 temporarily moving the bicarbonate balance to favour the formation of hydroxide ions. Unless the raw wastewater already has a high alkalinity, the only source of high pH in the pond is the effect of the algae.

Many authors argue that volatilisation is the dominant process for ammonia removal in waste stabilisation ponds (Reddy, 1983; Silva *et al.*, 1995; Shilton, 1996; Soares *et al.*, 1996). Shilton (1996) showed the rate of volatilisation has a positive correlation with the surface area of the pond and the initial concentration of ammonia in the water, and for very shallow ponds with high concentrations of ammonia, volatilisation can be significant. Reddy (1983) found that 54% of radiolabelled nitrogen unaccounted for in his algal system, and assumed lost through volatilisation or denitrification processes.

Pano and Middlebrooks (1982) proposed a model (equation 2.2) for the removal of ammonia from waste stabilisation ponds incorporating hydraulic loading, pH, temperature, and coefficients derived from empirical data.

$$C_e = C_o / \{1 + [(A/Q)(0.0038 + 0.000134\,T).\exp((1.041 + .044T)(pH - 6.6))]\} \quad (2.2)$$

where

C_e and C_o are the ammonia concentrations the pond effluent and influent respectively (mg/l N);
Q is the wastewater flow rate (m^3 / d)
A is the pond surface area (m^2) and
T is the temperature (^oC).

The equation assumes first order removal kinetics, complete mixing and temperatures up to 20^{o}C. Pano and Middlebrooks (1982) found a good fit with pond systems in temperate climates, though the model was developed using three systems which were extreme in their wastewater characteristics. The model was highly influenced by the data from Corrine, Utah which had very alkaline wastewater and a very high removal of ammonia; and by the data from Peterborough, New Hampshire, an overloaded system with poor removal and slightly acidic wastewater. The model does not include temperatures above 20^{o}C; the suggestion being that above this temperature stratification reduces the mean hydraulic retention time leading to a reduction in performance. Interestingly though, the Corinne data were not as well represented above this temperature as were the others used in the model. However, the model was successfully tested against four other systems, and Soares (1996) found good agreement with the model for data from ponds in north-east Brazil.

2.4.9.2 Ammonia assimilation

Algal uptake has been suggested as the major pathway for ammonia removal by Lai and Lam (1997) and Ferrara and Avci (1982). Ammonia is the preferred source of nitrogen for algae and has been shown to be taken up in preference to nitrate (which has to be reduced to ammonia before it can be assimilated) (Konig *et al.*, 1987; Mainer *et al.*, 2000). Algal uptake of ammonia was illustrated by tests on light and dark jars by Li *et al.* (1991): uptake was 87.5 % in a light jar and 3.2% in a dark jar with identical pH and temperature. However, the dry weight of nitrogen in algal cells is low (about 8% w/w) and cannot account for all the ammonia removal found in ponds (Green *et al.*, 1996; Lai and Lam, 1997). Reddy (1983) found that algae only assimilated 4.6% of radiolabelled $^{15}NH_4^+$.

Ammonia taken up by algae is converted to organic nitrogen which flows out or settles to the bottom. Anaerobic digestion of the sludge may reconvert it to ammonia which may feedback to the water column, though between 20-60% of the algal biomass is non-biodegradable and the nitrogen associated with this fraction is expected to remain in the sediment (Mara and Pearson, 1986). Excretion by grazing zooplankton is another pathway for the recycling of ammonia taken up by algae (Lai and Lam, 1997).

Ammonia assimilation into the algal biomass depends on the biological activity in the system and is affected by temperature, organic load, detention time, and wastewater characteristics (US Environmental Protection Agency, 1983). The optimum conditions for this process (i.e. high algal activity) are the same as for ammonia volatilisation. Ferrara and Avci (1982), who modelled nitrogen transformations in a facultative pond, found volatilisation to be a minor pathway, thus concluding that algal uptake was the major process.

2.4.9.3 Biological nitrification

Nitrification of ammonia is an aerobic chemoautotrophic process converting ammonia to nitrite and then nitrate. Evidence for nitrification is usually found by a decreasing ammonia concentration accompanied by an increasing nitrate concentration. There is little evidence for nitrification in facultative ponds because the concentrations of both nitrate and nitrite are usually very low in both inlets and outlets (Reed, 1985). If nitrification were occurring, some accumulation of nitrate would be expected in the surface aerobic layers where it could be formed. Nitrification is unlikely to occur in facultative ponds due to the low density of nitrifying bacteria found in the aerobic zone. This is thought to be due to the absence of physical attachment sites in the aerobic zone and possible inhibition by pond algae. The bacteria also tend to adsorb to particles and settle to the anoxic zone where nitrification is inhibited. It has been suggested that the low levels of nitrate found in ponds could be due to denitrification losses (anaerobic conversion of nitrate to nitrite to nitrogen gas) (Bucksteeg, 1987). Whilst denitrification can occur in the anoxic lower layers and sediments, or potentially during the night if anoxic conditions occur on the surface, some nitrate accumulation would still be expected. This is because, although many bacteria present in water are facultative denitrifiers, the rate of denitrification is very low in aquatic ecosystems (Horne, 1995). Also, evidence of nitrification, shown by increases in nitrate concentration, has been found on fully aerobic very shallow ponds (Reddy, 1983; Santos and Oliveira, 1987; Lai and Lam, 1997). It is thought, therefore, that nitrification is a minor pathway for ammonia removal in facultative ponds.

The algae play a key role in the removal of ammonia either by direct uptake, or by the creation of an environment for volatilisation (high pH) and nitrification (high DO). Although it is not certain as to the exact mechanism, the presence of algae has been observed to coincide with enhanced ammonia removal (Shillinglaw and Pieterse, 1977; US Environmental Protection Agency, 1983; Li *et al.*, 1991). At low concentrations, ammonium ions may adsorb to humus or inert particles and settle out (Green *et al.*, 1996) (Mainer *et al.*, 2000).

There is agreement that ammonia removal is improved by higher temperatures, high pH, increasing mean hydraulic retention time and reducing organic load (Lai and Lam, 1997; Muttamara and Puetpaiboon, 1997; Reed *et al.*, 1988). Chlorophyll-a and DO have also been suggested as important (Soares *et al.*, 1996; Lai and Lam, 1997) but they correlate with the other factors. It is difficult to distinguish between the possible ammonia removal mechanisms as they all respond to these factors.

2.4.9.4 Ammonia removal efficiency

Reported ammonia removal efficiencies vary greatly, between 0-95%, the higher values usually being associated with very long retention times (Mara and Pearson, 1986; Silva *et al.*, 1995). Many studies have shown that ammonia removal is between two to three times greater in summer than winter in temperate climates (Toms *et al.*, 1975; Pano and Middlebrooks, 1982; Bucksteeg, 1987; Santos and Oliveira, 1987; Racault *et al.*, 1995; Cauchie *et al.*, 2000). For example, the removal from the primary pond in the following systems in the US as reported by Pano and Middlebrooks (1982) were: Corinne, Utah: winter 62 %, summer 95%; Eudora, Kansas: 53% winter, 80% summer; Peterborough, New Hampshire: winter 0%, summer 38%. During the winter in cold climates facultative pond effluent ammonia may equal or exceed the influent concentration. Possible suggestions for this seasonal effect include: the loss or reduction of algal biomass (affecting uptake and volatilisation), low temperatures, and reduced hydraulic retention time due to increased rainfall during winter months.

In many countries there is increasing regulatory pressure for treatment systems to produce effluents with low ammonia concentrations. Attempts to modify the layout of ponds to improve ammonia removal include the inclusion of baffles or wetlands to provide nitrification sites (see Horne (1995) and Muttamara and Puetpaiboon (1997)) and the use of high rate algal ponds for enhanced ammonia uptake as described by Green *et al.* (1996).

2.5 Facultative pond design theory

Facultative ponds are primarily designed for BOD removal. Modelling of the processes within facultative ponds has been attempted to provide data on the correct loading to achieve the desired BOD removal and also to avoid pond failure.

According to the US EPA (1983), influent BOD, temperature and retention time are the principal parameters for the design of facultative ponds for BOD reduction. Temperature affects all metabolic processes and so is considered one of the most important factors. Sources of heat to the pond are solar radiation and influent wastewater (if it is at a higher temperature than the pond water). Cooling influences are evaporation, contact with cooler groundwater and wind action. Other important external factors affecting primary facultative pond processes as given by McGarry and Pescod (1970), are: depth, solar radiation intensity, and quality and quantity of influent waste. The wastewater has a profound effect on the design and even appropriateness of a waste stabilisation pond system. Reliable data are required on the chemical and physical characteristics and their variability, also reliable estimates of flow and its variability are required.

2.5.1 Design models

2.5.2 Hermann and Gloyna (1958)

Hermann and Gloyna (1958) developed an early kinetic theory to describe the behaviour of facultative ponds. With data from a series of model ponds they established that the retention time required for BOD removal is a function of temperature. Specifically, the retention time to give a 90% reduction in BOD_5 at 35^oC was 3.5 days. The retention time required at any other temperature was given by the following relationship:

$$\frac{R_T}{R_{35}} = q^{(35-T)} \qquad (2.3)$$

Where q = Arrhenius constant = 1.072
R_T is the retention time required for temperature T
R_{35} = 3.5 days

This model assumes that the influent BOD is 200 mg/l, thus a 90% removal results in effluent concentration of 20 mg/l in the coldest month. For any other value of BOD, the calculated retention time should be adjusted in the ratio BOD/200. The useful range is from 5^oC to about 35^oC, the lower limit resulting from the retardation of aerobic bacteria and algal activity. This simple design model implies that the removal during summer should be much higher than in winter, so does not take into account the effects of sludge feedback.

2.5.3 The Gloyna equation (Gloyna, 1976)

The Gloyna equation is an empirical model based on the experience of several hundred full-scale ponds, together with laboratory and pilot scale studies. These studies were conducted in the southwestern part of the US. The equation also takes an Arrhenius form and includes two toxicity factors for industrial or high sulphate waste.

$$R = 0.035 L_a \theta^{(35-T)} f . f' \qquad (2.4)$$

R = hydraulic retention time (d)
L_a = ultimate BOD influent (mg/l)
θ = Temperature coefficient = 1.085
T = pond water temperature in the coldest month (^oC)
f = algal toxicity factor = 1.0 for domestic
f' = sulphide or other direct chemical oxygen demand = 1.0 where sulphate < 500 mg /l

Equation 2.4 can be used to calculate the retention time required for an 80-90% removal of BOD (filtered effluent samples). It also shows that the retention time is a function of temperature, influent BOD concentration, algal toxicity and sulphide concentration. Gloyna suggested that the ultimate BOD be used in the equation because ponds usually have retention times of more than 5 days. Ultimate BOD values are not usually available and so sometimes the COD concentration is used or a multiplier (e.g. 1.5 x BOD_5). Again, the useful temperature range for the equation is between 5-35^{o}C.

From equation 2.4 it was intended that pond volume could be calculated as follows:

$$V = R\,Q \qquad (2.5)$$

where V = pond volume required (m^3) and Q = influent flow rate (m^3/d)

Gloyna stated that calculation of the pond area from his model should be based on an effective depth of 1 m, therefore, numerically V = A. However he recommended additional depth should be included in the pond to accommodate sludge accumulation depending on the climate as shown in Table 2.2

Table 2.2 The depth of ponds as suggested by Gloyna

Climate	depth (m)
uniform temperature : tropical to subtropical low settleable solids	1
as above but with modest settleable solids	1.25
significant seasonal variation in temperature	1.5
severe climates	1.5-2

The Gloyna equation assumes adequate sunlight (as that found in the study area), but this difference may be factored into the equation by adjusting the calculated pond volume by the ratio of the sunlight in the application area to that found in the SW of the United States. When the equation was evaluated against pond systems in the temperate zone of the US, equation 2.6 was shown to give the best fit (Reed *et al*., 1988):

$$V = 0.035\ Q\ C_o\ (1.099)^{LIGHT\ (35-T)\ /250} \qquad (2.6)$$

C_o = BOD_5 in the influent (mg/l)
LIGHT = solar radiation, (cal. cm^{-2})

The Gloyna equation provides an estimate of total area required for BOD removal but does not include the area requirements for the primary facultative pond (or first cell).

2.5.4 Marais and Shaw (Complete mix) model (Marais and Shaw, 1961)

Marais and Shaw proposed a model based on first order kinetics in a complete mix reactor. In a complete mix situation, the in-pond BOD concentration is constant and equal to the effluent BOD concentration:

$$\frac{C_n}{C_o} = \left[\frac{1}{1 + k_c r_n}\right]^n \qquad (2.7)$$

C_n = effluent BOD_5 from pond n (mg/l)
C_o = influent BOD_5 (mg/l)
k_c = first order reaction rate constant (/d)

r_n = hydraulic retention time in each pond (d)
n = number of ponds in series

The value of k_c is temperature dependent as follows:

$$k_{C_T} = k_{C_{35}}(1.085)^{T-35} \tag{2.8}$$

where k_{C_T} = reaction rate at minimum operating water temperature
$k_{C_{35}}$ = reaction rate at 35^{o}C = 1.2 /d
T = minimum operating water temperature (^{o}C)

The equation may be used to determine both the retention time and the number of ponds required.
Marais and Shaw proposed that the maximum BOD_5 in primary cells ($(C_e)_{max}$) to avoid anaerobic conditions and odours should be 55 mg /l. The retention time required for the first pond may be calculated by substituting $(C_e)_{max}$ for C_n in equation (2.7) where n=1.

The permissible depth for the pond is given by:

$$(C_e)_{max} = \frac{700}{2d+8} \tag{2.9}$$

where d = the permissible depth of the pond (m). Using this equation, when $(C_e)_{max}$ = 55 mg/l, d = 2.4 m. The weakness in the model lies in this calculation of depth because an assumption of light penetration to the pond bottom is made and this is unrealistic for turbid primary facultative ponds. Mara (1976) pointed out that the equation is most sensitive to the numerator (700: whose selection is arbitrary) than to *d*. In a worked example given by US EPA (1983), the model produced a surface loading rate on the first cell of 135 kg BOD / ha.d at 0.5^{o}C. This was as a result of the depth of 2.4 m that is used in the derivation of area. If a depth of 1 m were used (as Gloyna suggested), the surface loading would be 2.4 times less (i.e. 56 kg/ha.d at 0.5^{o}C).

2.5.5 Plug flow

A plug flow system is a theoretical concept where every element of the flow leaves the pond in the same order in which it entered, thus every element is exposed to treatment for the same length of time (equal to the theoretical hydraulic retention time). A plug flow model, as equation 2.10, has been shown to best describe the performance of ponds in series (Reed *et al.*, 1988).

$$\frac{C_e}{C_o} = e^{-k_p R} \qquad (2.10)$$

C_e = effluent BOD_5 concentration (mg/l)
C_o = influent BOD_5 concentration (mg/l)
k_p = plug flow first order reaction rate (/ d)
R = hydraulic retention time (d)

k_p is related to temperature as follows:

$$k_{p_T} = k_{p_{20}}(1.09)^{T-20} \qquad (2.11)$$

where

k_{p_T} = reaction rate at temperature T
k_{p20} = reaction rate at 20^oC and
T = minimum operating water temperature

k_{p20} depends on the BOD_5 surface loading rate as shown in Table 2.3, but if this is not known, Reed *et al.* (1988) suggest that a value of 0.1 /d may be used.

Table 2.3 $k_{p\ 20}$ values for different surface loadings

kg/ha.d	k_{p20} /d
22	0.045
45	0.071
67	0.083
90	0.096
112	0.129

The plug flow model can be used to calculate the retention time required for specified BOD removal requirements. If the flow rate is known, the required volume may be calculated using eqn. 2.5 and hence area for a specified depth.

2.5.6 Dispersed flow

The Wehner-Wilhelm equation (eqn. 2.12) for arbitrary flow was proposed by Thirumurthi (1969) as a method to design facultative ponds.

$$\frac{C_e}{C_o} = \frac{4ae^{\frac{1}{2D}}}{(1+a)^2 e^{\frac{a}{2D}} - (1-a)^2 e^{\frac{-a}{2D}}} \tag{2.12}$$

C_e = effluent BOD_5 concentration mg/l
C_o = influent BOD_5 concentration mg/l
a = $\sqrt{1+kRD}$
k = first order reaction rate (/d)
R = hydraulic retention time (d)
D = dispersion number = $\frac{H}{vl} = \frac{Ht}{l^2}$
H = axial dispersion coefficient (area/time)
v = fluid velocity, (length/time)
l = length of travel path of a typical particle

$k_T = k_{20}(1.09)^{T-20}$
k_T = reaction rate at minimum operating temperature
k_{20}= reaction rate at 20°C = 0.15 /d
T= minimum water temperature (°C)

Thirumurthi developed a chart to facilitate the use of the complicated equation where kt is plotted against the % BOD remaining in the effluent for varying dispersion numbers (D) varying from zero for a plug flow reactor to infinity for a complete mix reactor. Dispersion numbers for ponds range from 0.1 to 2 with most values not exceeding 1.0. Using the equation is difficult because the two constants, k_t and D must be selected correctly. Once they are selected, the equation may be solved using the chart or else by trial and error: substituting different values of t until there is agreement on both sides of the equation. Due to the difficulty of selecting k_t and D, design using one of the simpler models has proven to be just as satisfactory (Reed *et al.*, 1988).

2.6 Surface BOD loading

The surface BOD loading approach is purely empirical, using data from existing ponds in a particular region. The basis of the approach is that for every climate there is an appropriate value of λ_s: the surface BOD load (kg BOD/ha.d) which may be applied to a pond for a given removal efficiency or before pond failure. Using influent flow and BOD concentration data, the area required for the first and subsequent ponds may be calculated.

The surface BOD loading rate has come into widespread use because it directly determines the land requirements which in turn determine the major proportion of the construction costs of ponds. Most of the processes in a facultative pond are surface related phenomena: sunlight reception, algal growth, wind aeration, etc. and it has been found that surface loading values give closer correlation with performance data than volumetric loading values (McGarry and Pescod, 1970). Because the pond volume is not relevant to this approach, the depth of the pond may be decided by other factors (e.g. sludge storage capacity). It is generally accepted that this approach should only be used if enough local field data is available. However, there have been attempts to use the values determined from field data in various climates to provide global models to be used as a guide in the absence of local data, for example the work by McGarry and Pescod published in 1970.

2.6.1 McGarry and Pescod empirical procedure (McGarry and Pescod(1970))

McGarry and Pescod collated data from 143 different climatic[6] conditions and reported that BOD removal in primary facultative ponds was between 70-90%. On statistical modelling of the data, they found that pond performance (expressed as surface BOD removal) was related only to surface BOD loading as follows:

$$\lambda_r = 10.75 + 0.725\,\lambda_s \qquad (2.13)$$

λ_r= surface BOD removal (kg/ha.d)

The regression line had a very high correlation coefficient (0.995) suggesting that BOD removal might be independent of all other factors including temperature. However, it was also found that the BOD loading data correlated with temperature: i.e. higher loadings were used in locations with higher temperatures. They were unable to establish a correlation between hydraulic retention time and removal efficiency, even when partitioned for temperature. Other similar relationships have been found, for example in Portugal by Gomes de Sousa (1987).

2.6.2 McGarry and Pescod: Envelope of failure

Using data on pond failures in different climates, McGarry and Pescod developed a mathematical model of surface BOD loading to point of failure that includes temperature as the only explanatory variable (equation 2.14a).

$$\lambda_S=11.2\,(1.054)^T \qquad (2.14a)$$

where
λ_S = the maximum BOD_5 loading before failure (kg/ha.d)
T= temperature in ^oF

or $\lambda_S = 60\,(1.099)^T$ where T= temperature in ^oC (2.14b)

[6] Climate was defined in terms as the mean monthly ambient temperature (representing both pond water temperature and incident solar radiation).

The boundary curve of equation 2.14a is said to approximate to operation when the algal layer is just capable of maintaining aerobic conditions at any temperature and solar radiation. It is an Arrhenius type equation and was considered to reflect oxygen production by algae as a function of temperature (though the equation is empirical in origin). This assumption excludes the effects of wind mixing. The curve extends to cover temperatures up to 30^{o}C, though there are no failure points in their data above 5^{o}C.

2.6.3 Surface loading: derivatives of McGarry and Pescod

McGarry and Pescod's failure model was adapted by Mara and Pearson (1987) by incorporating a safety factor to give a global design equation for facultative ponds as follows:

$$\lambda_S = 350\,(1.107 - 0.002T)^{T-25} \qquad (2.15)$$

where T= temperature in ^{o}C (for T = 8-25 ^{o}C)[7]

Secondary facultative ponds receive pre-settled influent therefore a higher proportion of the influent BOD enters the pond liquid than for a primary pond. Consequently, it was recommended that secondary facultative ponds should receive 30% less loading than primary ponds (Mara and Pearson, 1986).

Other similar relationships to have been derived, though mainly for warm climates, for example:

$\lambda_S = 20\,T - 120$(Mara, 1976)

$\lambda_S = 20\,T - 60$ (Arthur, 1983)

$\lambda_S = 10\,T$ (Mara and Pearson, 1998) for Israel

$\lambda_S = 357.4\,(1.085)^{T-20}$ (Bartone, 1985) for Peru

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[7] for temperatures below 8^{o}C, an surface loading of 80 kg/ha.d is recommended
Mara, D. D. and Pearson, H. (1998). *Design Manual for Waste Stabilization Ponds in Mediterranean Countries,* Leeds: Lagoon Technology International Ltd..

2.6.4 Indian empirical procedure (CPHERI, 1970)

Experience of pond operation in India yielded a design equation which relates the permissible loading to latitude as follows:

$\lambda_S = 375 - 6.25\ (\mathrm{Lat})$ (2.16)

where Lat = latitude (range in India 8-36^o N) (*range in UK 50-60^o N)*

2.6.5 Surface loading method: interpretation of local data

Problems can be encountered in the interpretation of surface loading data; one of the main pitfalls is the inconsistent use of units. The unit of kg BOD/ha.d is directly quantifiable: relating BOD load to pond area; however, design standards frequently use a population equivalent: m^2/person. The latter unit is useful for sizing ponds in the absence of wastewater quality data, but awkward for comparing design standards and pond performance from place to place because an assumption of the BOD contribution per person per day must be made and this also varies from place to place. For example, CEMAGREF (1997) assume 35-40 g BOD/person d in rural areas of France, Mara and Pearson (1998) use 50g BOD/person.d, EU standards assume 60g BOD/person.d (Council of the European Communities, 1991) and in New Zealand 70g BOD/person.d is assumed (84 kg ≡ 1200 persons) (Mara, 1995). These assumptions make a significant difference to the calculated surface BOD loading as Table 2.4 illustrates.

Wherever design requirements are expressed as m^2/person there is the possibility of confusion in the interpretation of data leading to disagreement between design standards.

Table 2.4 Facultative pond loading conversion

m^2/ person	**kg BOD/ha . d**			
	assuming 40g BOD per person per day	assuming 50g BOD per person per day	assuming 60g BOD per person per day	assuming 70g BOD per person per day
1	400	500	600	700
2	200	250	300	350
3	133	167	200	233
4	100	125	150	175
5	80	100	120	140
6	67	83	100	116

PART II: Waste Stabilisation Pond Experience in Temperate Regions

2.7 Introduction

A generic classification of world climate, proposed by Strahler (1969), divided the world into 14 climatic regions. In this classification, the UK climate is no.7: "middle latitude - marine west coast" which covers areas between 40-60^{o}N and S latitudes with temperate rainy climates and short cool summers. Other areas in the world thus classified are: Western Europe (France, Germany, Eire, Belgium); NW coast of North America, Southern Chile, Tasmania and New Zealand. In the absence of local data, the experience of waste stabilisation ponds in these areas should give the next best guide to pond performance in the UK.

2.8 United States

Prior to 1950, waste stabilisation ponds were rare and their use generally discouraged in the United States. However, by 1960 there were 469 systems, by 1970 there were 1743 systems (Van Heuvelen, 1970) and by 1989 there were 7607 (Water Environment Federation, 1998) operating systems. Designed in series with a minimum of 3 ponds, systems in the United States are typically 1.5-2.5m in depth with a retention time of between 5-30 days per pond. In 1960, the Missouri River Basin Engineering Health Council committee published design criteria for the ten-state area. The 'Ten State Standards' were incorporated into the 1983 United States Environmental Protection Agency (US EPA) Design Manual: "*Municipal Wastewater Stabilization Ponds*" which contains the standards applied in the United States today, though individual states may have their own standards (Water Environment Federation, 1998; Nelson, 2000). The surface loading criteria are given in Table 2.5.

Table 2.5 Summary of pond surface BOD loading as recommended by the United States EPA manual

Average Winter Air Temperature ^oC	BOD_5 loading (kg/ha.d) across whole system	BOD_5 loading (kg/ha.d) in first cell*
> 15	**45-90**	**100**
0-15	**22-45**	*70***
< 0	**11-22**	**40**

*to avoid anaerobic conditions and odours
**an interpolated average, and not stated by the US EPA

These design standards were influenced by the work done by Neel *et al.* (1961) on experimental ponds in Missouri for one year during 1957-58. There were five ponds, each with an area of 0.3 ha and 0.762 m in depth, loaded at 20, 45, 67, 90 and 112 kg BOD/ha. d. The researchers found that the ponds loaded at >45 kg/ha.d experienced odour problems, zero DO and low chlorophyll-a concentrations, after the ice cover melted in February 1958. They hypothesed that at loadings above 45 kg/ha.d, the rate of oxygen consumption was too high for the photosynthetic rate permitted by light penetration through ice. The ice cover for these ponds lasted for 36 days.

The work on ponds in the United States is very comprehensive, but perhaps should only be applied to the UK with caution, as the climate over most of the United States is continental in nature.

2.9 New Zealand

Waste stabilisation ponds are the most common method of wastewater treatment in New Zealand: in 1987, 63% of all sewered communities of >1000 people were using them. Systems range in size from 0.1-500 ha. Most systems were constructed between 1960 and 1985 and have either one pond or two of equal size in series (Archer and Mara, 2002). The 1974 Ministry of Works and Development recommendations are: two ponds in series, the first (facultative) pond should have a maximum surface BOD loading of 84 kg/ha.d and the second (maturation) pond should provide a retention time of at least 20 days (Ministry of Works and Development, 1974). Although the ponds offered adequate BOD and SS removal, higher levels of treatment are now required for nutrients and pathogens. Since 1995, pond systems with between 4-6 ponds (including maturation ponds) have been constructed or retrofitted.

2.10 France

The first waste stabilisation pond system was installed in France in 1965 (Bountin *et al.*, 1987). Their use developed rapidly: 115 systems in 1980, 1289 in 1986, and over 2000 by 1992 (Racault *et al.*, 1995). The vast majority of these systems are for the treatment of domestic sewage from small rural communities of average size 600 p.e. The majority of French waste stabilisation pond systems consist of a series of three ponds covering a total specific area of 10 m^2/p.e. The first pond takes up 50% of the total surface area, this is to permit an aerobic surface layer and prevent noxious odours (Racault, 1993). Assuming a BOD contribution of 50 g / person .d, the permissible loading was originally 50 kg BOD /ha .d for the whole system and 100 kg BOD/ha.d for the first pond. The average retention time is 60 days. Recently, the recommendations have been revised by CEMAGREF (1997) in the light of operational experience. They now recommend an area of 6 m^2/ person for a primary facultative pond, the equivalent of 83 kg/ha.d, and 11 m^2/ person overall.

Racault *et al.* (1995) collected data from 178 systems in their 1992 survey of French ponds; the survey included systems with a primary facultative pond followed by one or more maturation ponds; for many of the systems only one set of diurnal data was available. They discovered that the average actual loading applied to the systems was 25.5 kg BOD/ha.d (50% of the recommended loading), and that 70 % of the systems met the French discharge standards for BOD, COD and SS. The average removals for BOD and SS were 85% and 77% respectively, though there was large dispersion in the data. Removals for ammonia and phosphorus were 70% and 60%, respectively, on average. No seasonal difference was found in BOD or SS removal, though ammonia removal was found to be very seasonal, with effluent concentration three times higher in winter than summer. A correlation was found between effluent ammonia concentration and organic surface loading in winter, but not in summer. They concluded that filtration systems may be needed to upgrade the effluents to meet tighter EC standards in future.

2.11 Germany

In his review of German experiences with ponds, Bucksteeg (1987) noted that there were more than 1000 waste stabilisation ponds in Germany, all in rural areas for population equivalents of up to 1000. In the southern part of Germany the typical layout consists of an anaerobic pond of specific volume 0.5-1.0 m^3/person, followed by two equally sized ponds, each with a specific area of 2.5-5 m^2 / person. In the northern part of Germany, the typical layout is three ponds in series covering a total specific area of 10-15 m^2/ person, distributed in the ratio 3:4:3. Thus the specific area assigned to the first pond is 3-5 m^2 /person, or 100-167 kg/ha.d (assuming 50 g BOD / person.d).

Bucksteeg summarised the performance of the systems used in southern Germany, but indicated that similar results had been obtained from the northern systems. BOD effluent quality did not vary seasonally, and was related to surface organic load. Total specific areas of 5 m^2/person and 10 m^2/person led to an effluent quality of <35 mg/l BOD and <20 mg/l BOD, respectively. Increasing the specific area above 10 m^2/ person did not lead to a better quality effluent. Ponds with a specific area of >5 m^2/ person were capable of taking hydraulic shock loads of up to 40 times dry weather flow with no effect on performance. Ammonia removal was found to be very seasonal and sensitive to increases in surface organic loading. Effluent quality for summer was between 1-23 mg/l NH_3-N and for winter, 7-33 mg/l NH_3-N. On average, a total specific area of 15 m^2/ person gave a <10 mg/l ammonia-N effluent concentration, whilst 10 m^2/ person gave <15 mg/l. Phosphorus reduction was also seasonal: 75% reduction in summer, 19% in winter for orthophosphate; and 63% in summer and 19% in winter for total phosphorus. A specific pond area of >10 m^2/person was required for an effluent concentration of <6 mg/l of phosphate-P.

2.12 Belgium

The use of ponds in Belgium has been documented in the Wallonne region. In 1986, there was only one waste stabilisation pond system in this region and 4 under construction; design area for the facultative pond was 5-11 m^2 /person (Vuillot and Boutin, 1987). By 1994, there were 49 systems, of which 20 were unassisted and 9 of these were for full treatment (Godeaux *et al.*, 2000). The systems for full treatment, typically sized for 10 m^2/person (50 kg/ha.d), were usually organically underloaded and hydraulically overloaded as a result of the very dilute wastewater found in the rural areas where the ponds were located. The effluent limits for BOD and SS of 25 mg/l and 35 mg/l, were achieved in 80% of samples (Godeaux *et al.*, 2000).

2.13 Denmark

From the 1940's, waste stabilisation ponds were developed in Denmark for secondary treatment of domestic or industrial waste (Vuillot and Boutin, 1987). In 1986 there were 66 plants treating population equivalents of between 50-3000, though most were between 500-1000. The pond systems typically consisted of an anaerobic pond followed by secondary facultative pond/s. The design volumetric loading applied to the anaerobic pond was 0.9 m^3/ person and the surface loading to the remaining area was 8-10 m^2 / person (75-60 kg/ha.d). The ponds were unable to meet the new, tighter nutrient discharge standards (8 mg/l N and 1 mg/ l P) especially during the winter. By 1986, all existing waste stabilisation ponds had been upgraded with either biological and / or chemical treatment.

2.14 Summary

The data from countries with a similar climate to the UK suggest that the appropriate surface BOD loading for a primary facultative pond is between 48-167 kg/ha.d (depending on the BOD/ person assumption). Performance data from the extensive use in France and Germany suggest that waste stabilisation pond systems should operate effectively for BOD removal in the UK climate at all times of the year, though seasonal effects on nutrient removal may be a concern.

PART III Waste Stabilisation Ponds in the United Kingdom

2.15 Introduction

In the UK pond systems have been used for a variety of purposes including tertiary treatment, grey water treatment, and the treatment of raw sewage. In comparison to other low-cost sewage treatment methods there has been relatively little research published on their design or performance. Mara *et al.* (1998) reported 19 waste stabilisation pond systems in the UK, as given in Table 2.6.

2.16 Rye Mead maturation pond system

The first extensive study published on waste stabilisation ponds in the UK was from the maturation pond system at Rye Meads in Hertfordshire. These ponds, used for the tertiary treatment of activated sludge effluent, were studied between 1971-75 and the results published by Potten (1972) and Toms *et al.* (1975). Although on maturation ponds, the work is comprehensive and has some useful data on the possible performance of waste stabilisation pond systems in the UK climate.

The pond system consisted of three maturation ponds in series, with retention times of between 3.6 - 9.3 days. The authors found that the ponds remained aerobic and produced an effluent with reduced concentrations of BOD, SS, and pathogenic organisms, as compared to the activated sludge plant effluent. The removal of BOD and SS was found to be incompatible with the removal of other parameters such as ammonia and phosphate due to the growth of algae. Although appreciable removals of ammonia, nitrate and phosphate were recorded, there was a great decrease in the reduction of these nutrients during the winter months. It was noticed that the greatest reduction of nutrients occurred in the ponds with the longest overall retention time.

Table 2.6 Waste stabilisation ponds in the UK (Mara *et al.*; 1998)

Location	Year Commissioned	No. of ponds in series	Effluent quality requirements (mg/l)
Sturts Farm, Dorset	1989	3[a]	BOD 20; SS 30; amm.N 20
Tigh Mor Trossachs, Perthshire	1992	3	BOD 20; SS 30; total P 3
Larchfield/1, Teesside	1993	3	BOD 40; SS 60;
Hapstead House, Devon	1993	3[a]	BOD 40; SS 60;
Dulo Manor, Cornwall	1994	3[b]	Winter: BOD 10;SS 10; amm.N 5 Summer: BOD 5; SS 5; amm.N 2
Scolton Manor, Pembrokeshire	1994	3	None (discharge to groundwater)
Nature's World, Teesside	1994	3	None (volume consent)
Acklam Grange School, Teesside	1994	3	None (volume consent)
Larchfield/ 2, Teesside	1995	3[a]	BOD 40; SS 60
Combermere, Shropshire	1995	3	None (discharge to planted leachfield)
Corfe Castle, Dorset	1996	3	None (discharge to groundwater)
Drummuir Castle, Banffshire	1995	3	BOD 10; SS 10
Barnardiston School, Essex	1995	2	BOD 20; SS 30;
Gordonstoun School, Morayshire	1995	2[c]	BOD 10; SS 10; amm.N 10
Burwarton Estate, Shropshire	1995	3	BOD 25; SS 45; amm.N 10
Botton Village, North Yorkshire	1997	3	BOD 40; SS 30
Thornage Hall, Norfolk	1997	3	BOD 40;SS 60; amm.N 10
Earth Balance, Northumberland	1997	3	BOD 40; SS 60
Newnham, Gloucestershire	1997	2	BOD 40; SS 60

[a] Pretreatment in a septic tank
[b] The WSP are followed by a soakaway.
[c] Grey water only.

2.17 Burwarton Estate waste stabilisation pond system

The waste stabilisation pond system at Burwarton Estate, Shropshire was constructed in 1995 and serves a population of 150. The plant comprises a primary facultative pond and two maturation ponds in series. Mara *et al.* (1998) calculated that the design loading to the first pond is 55 kg BOD/ha.d, and the total surface area of the ponds is 23 m^2/ person. A rock filter surrounds the outlet of each pond and the final pond effluent passes through a leachfield before discharging to a watercourse; in dry conditions the leachfield absorbs all the flow. The summer performance of the system was reported by Cogman (1995), Schembri (1996) and Mara *et al.* (1998). According to Mara *et al.* (1998), the effluent from the second maturation pond was consistently achieving <20 mg/l BOD, <30 mg/l SS, and ammonia-N concentrations of between 1 - 3 mg/l. Removal rates varied between 80-95% BOD, 64-95% SS, and 86%-99% ammonia-N. It was observed that the majority of organic matter removal occurred in the facultative pond, whilst a large degree of ammonia removal occurred across the maturation ponds.

2.18 Sturts Farm pond system

The Sturts Farm System was the first documented pond system in the UK incorporating a facultative pond. Commissioned in 1989, the system consists of a septic tank followed by three ponds in series, each with a horizontal-flow reed bed at the outlet. The performance of the system was investigated by Mainwaring (1991) in July 1990, September 1990 and February 1991. The surface loading on the first pond was around 75 kg/ha.d (based on influent BOD 150 mg/l and 11 m^3/d flow). Mainwaring found the system had an average removal of 90% BOD, 88% SS, 93 % ammonia and 63 % phosphate. In July, SS rose in the first pond due to algae, but had reduced to an average of 6 mg/l after the second pond (95% removal). In September, BOD removal in the first pond was between 81-95 %. Winter monitoring revealed good removal for BOD: 86% in the first pond, 94 % overall; and SS: 60% in the first pond, 96% overall. The summer monitoring for ammonia revealed a removal of 70% in Pond 1 and 98% in Pond 2. However, in February there was no removal of ammonia in pond 1; 10% removal in Pond 2, and 92 % removal in pond 3. Phosphate removal was between 62-65% across the system in both summer and winter. He concluded therefore, that there was good removal for BOD and SS in both winter and summer conditions, though ammonia removal was inadequate during the winter. Phosphate removal was accelerated in the presence of algae, but otherwise would require chemical precipitation. Mainwaring hypothesised that nitrification was occurring in the ponds during summer, followed by denitrification in the reed bed. This conclusion was purely based on the observation of low ammonia and nitrate concentrations in the effluent.

2.19 Wick St Lawrence (assisted lagoons)

Wessex Water undertook a pilot scheme between 1993-94 to investigate the performance of aerated facultative ponds using wind-powered aerators at the Wick St Lawrence wastewater treatment plant near Bristol. The aerators are manufactured by an American based company called Lake Aid Systems (LAS) International. Existing storm tanks were used to provide a series of three ponds, each with its own wind-powered aerator. The system was fed with screened sewage from a population equivalent of 250, with a design flow of 43 m^3/ day and an average influent BOD_5 of between 140-350 mg/l. The loading on the first cell was between 60-150 kg BOD_5/ha.day. The average removals from the first cell between December and July were 84-95% BOD; 53-92% SS; 0-95% ammonia-N (Lake Aid Systems International Ltd, 1998). The concentrations of BOD and SS in the effluent remained fairly constant throughout the sampling period. However, from mid-1994, concentrations of ammonia rose steadily until they averaged about 11 mg/l. On occasions during the summer months algal blooms would cause an increase in SS in the effluent.

2.20 Errol aerated lagoons

A full-scale wind-aerated lagoon system was commissioned in July 2001 at Errol, near Dundee by North of Scotland Water Authority. Built for a population of 2000, the system has two ponds, each 0.541 ha with an operating depth of 3.35m; the surface loading on the first pond is approximately 200 kg BOD_5 / ha.d (Wynnes, 2002). The primary pond has two wind aerators with back-up diffused aeration using a power consumption of 11 kW /d. (Lovelace, 2002). Between September-October 2001, the average removal across the whole system was 94% and 92% for BOD and SS respectively (Lake Aid Systems International Ltd, 2001).

2.21 Summary

The experience of waste stabilisation ponds in the UK suggests that satisfactory removal of nutrients may be a problem during the winter. The work carried out on the Burwarton waste stabilisation pond system indicates that ponds can operate efficiently in the UK during the summer (Mara *et al.* 1998). In this system, the removal of nitrogen appears to be efficient, however, this system is much larger than its French and German equivalents.

2.22 UK temperature

The UK mean annual air temperature at low altitudes varies between 8.5^{o}C to 11^{o}C, decreasing by about 0.5^{o}C for every 100m increase in altitude. The UK has a maritime climate with very mild winters for its latitude position. The winter air temperature is influenced by the surface temperature of the surrounding sea; the coldest month is January. The critical period for pond design is asserted to be the coldest month. Between 1961 and 1990, the average January air temperature across the UK was between –1^{o}C - +6^{o}C; for most of the land area it was between 2-4^{o}C. Average temperatures of 1^{o}C or less were found only on very high ground, and of 5^{o}C or more only in the south and westerly coastal regions (The Met Office UK, 2000).

2.23 Application of surface loading data and models to UK conditions

Assuming an average air temperature range of 2-4^{o}C during the coldest month, Table 2.7 summarises the surface BOD loadings potentially suitable for the UK as suggested by the various models and recommendations of other countries.

Table 2.7 Possible surface loading values for the UK

Source of data	equation / assumption	loading range for UK facultative ponds (kg/ha.d)
McGarry and Pescod Mara	$\lambda_S = 60\ (1.099)^T$ For temperatures < 8^oC	72-88 80
Arthur	20T-60	0-100
Indian empirical	375-6.25 L (for 50^oN)	0-62
United States EPA	for $2\text{-}4^oC$ (interpolated from $0\text{-}15^oC$ range)	48-56
France	50 g BOD / person.d	83
New Zealand	70 g BOD / person.d	84
Northern Germany	50 g BOD / person.d	100-167

The table shows that there is a consensus between McGarry and Pescod, Mara, France and New Zealand on the surface loading required (approximately 80 kg/ha.d). The range given by Arthur and the Indian procedure were developed for warmer climates and are not sufficiently precise. The biggest disagreement is between the German and United States' recommendations. These inconsistencies may be due to differences in climate, culture, or expectations of wastewater treatment systems. However, they may simply be due to the differences in calculating loading or the use of terminology.

2.24 Conclusion

The influence of climate on the performance of waste stabilisation ponds has been well documented and, although the strongest influence has been identified as relating to temperature, many other climatic factors play a part. Every region on Earth has a unique combination of climatic factors which affects pond performance; for this reason local field data collection is essential.