Development Document for the Proposed Effluent Limitations Guidelines and Standards for the Meat and Poultry Products Industry Point Source Category (40 CFR 432)

EPA-821-B-01-007

January 2002

U.S. Environmental Protection Agency
Office of Water (4303T)
Washington, DC 20460

Complete proposed document available at:
http://www.epa.gov/ost/guide/mpp/
The Final Development Document is available as well.

SECTION 8

WASTEWATER TREATMENT TECHNOLOGIES AND POLLUTION PREVENTION PRACTICES

8.1 INTRODUCTION

This section describes unit processes that are currently in use or may be used to treat meat and poultry products (MPP) wastewaters. A variety of unit processes are used to provide primary, secondary, and tertiary wastewater treatment; however, because of the similarities in the physical and chemical characteristics of meat and poultry products wastewaters, EPA identified no practical difference in the types of treatment technologies between meat and poultry products facilities (e.g., primary treatment for removal of solids, biological treatment for removal of organic and nutrient pollutants). In addition, the unit processes that are used in the treatment of MPP wastewaters are similar to those normally used in the treatment of domestic wastewaters (Eremektar et al., 1999; Johnston, 2001). In this section, those unit processes most commonly used or potentially transferable from other industries for the treatment of MPP wastewaters are described and typical combinations of unit processes are outlined.

Wastewater treatment falls into three main categories: (1) primary treatment (e.g., removal of floating and settleable solids); (2) secondary treatment (e.g., removal of most organic matter); and (3) tertiary treatment (e.g., removal of nitrogen or phosphorus or suspended solids or some combination thereof). MPP facilities that discharge to a publicly owned treatment works (POTW), typically employ only primary treatment; however, some facilities may also provide secondary treatment, as demonstrated in the data provided in the MPP detailed survey. MPP facilities that discharge directly to navigable waters under the authority of a National Pollutant Discharge Elimination System (NPDES) typically both primary and secondary treatment to generated wastewaters. As also described in the MPP detailed surveys, many direct dischargers also apply tertiary treatment to wastewater discharged under the NPDES permit system. Table 8-1 identifies the types of wastewater treatment commonly found in the MPP industry.

Table 8–1. Distribution of Wastewater Treatment Units In MPP Industry

Treatment Category	Treatment Unit	Percent of Direct/Indirect Discharging Facilities Having The Treatment Unit In Place	
		Direct Discharger	Indirect Discharger
Primary treatment	Screen	98 percent	64 percent
	Oil and Grease Removal	83 percent	77 percent
	Dissolved Air Floatation	81 percent	46 percent
	Flow Equalization	75 percent	34 percent
Secondary and tertiary treatment	Biological Treatment [a]	100 percent	13 percent
	Filtration	23 percent	0 percent
	Disinfection	92 percent	0 percent

[a] Biological treatment includes any combination of the following: aerobic lagoon, anaerobic lagoon, facultative lagoon, any activated sludge process, and/or other biological treatment processes (e.g., trickling filter).
Source: EPA Detailed Survey Data.

8.2 PRIMARY TREATMENT

As noted above, primary treatment involves removal of floating and settleable solids. In MPP wastewaters, the typical unit processes used for primary treatment are screening, catch basin, dissolved air flotation (DAF), and flow equalization. Chemicals are often added to improve the performance of the treatment units (e.g., flocculant or polymer addition to DAF units). Primary treatment has two objectives in the MPP industry: (1) reduction of suspended solids and biochemical oxygen demand (BOD) loads to subsequent unit processes; and (2) the recovery of materials that can be converted into marketable products through rendering.

8.2.1 Screening

Screening is typically the first and most inexpensive form of primary treatment. Screening removes large solid particles from the waste stream that could otherwise damage or interfere with downstream equipment and treatment processes, including pumps, pump inlets, and pipelines (Nielsen, 1996). There are several types of screens used in wastewater treatment including: (1) static or stationary, (2) rotary drum, (3) brushed, and (4) vibrating. Static, vibrating, or rotary drum screens are most commonly used as primary treatment (USEPA, 1974, 1975). These screens use stainless steel wedge wire as the screen material and remove medium and coarse particles between 0.01 to 0.06 inches in diameter. Generally, all wastewater generated in MPP facilities is screened before discharge to subsequent treatment processes. Use of screens aids in recovery of valuable by-products that are sometimes used as a raw material for the rendering industry and subsequent industries (Banks and Adebowale, 1991; USEPA, 1974; USEPA, 1975). The use of secondary screens is becoming more prevalent in the industry. Secondary screening has the advantage of by-product recovery prior to adulteration by coagulants and reduces the volume of solids to be recovered in subsequent unit processes, such as the dissolved air flotation (Starkey and Wright, 1997).

The following describes the main types of screens used at MPP facilities.

8.2.1.1 Static Screens

The primary function of a static screen is to remove large solid particles (USEPA, 1974; USEPA, 1975). For example, the physical nature of slaughterhouse raw wastewater can include coarse, suspended matter (larger than 1 mm mesh) which is insoluble, slowly biodegradable, and 40 to 50 percent of the raw wastewater COD (Johns, 1995). Screening can be accomplished in several ways, and in older versions, only gravity drainage is involved. A concavely curved screen design using high-velocity pressure feeding originally developed for mineral classification has been adapted to meet MPP wastewater treatment needs. This design employs bar interference to the slurry, which knives off thin layers of the flow over the curved surface. The screen material usually is 316 stainless steel although harder, wear-resistant stainless alloys may also be used for special purposes. Openings of 0.025 to 0.15 cm (0.01 to 0.06 inch) meet normal screening needs (USEPA, 1974; USEPA, 1975). Figure 8-1 shows a general schematic of a static screen.

In some poultry products facilities, “follow-up” stationary screens, consisting of two, three, and four units placed vertically in the effluent sewer before discharge to the municipal sewer, have successfully prevented escape of feathers and solids from the drains in the flow-away screen room and other drains on the premises. These stationary “channel” screens are framed and are usually constructed of mesh or perforated stainless steel with ¼- to ½ -inch openings. The series arrangement permits removal of a single screen for cleaning and improves efficiency. The three-slope static screen is being used in a few poultry products facilities as primary treatment (USEPA, 1975). Static screens can be used in series to remove of coarse particles first before further screening by finer mesh screens.

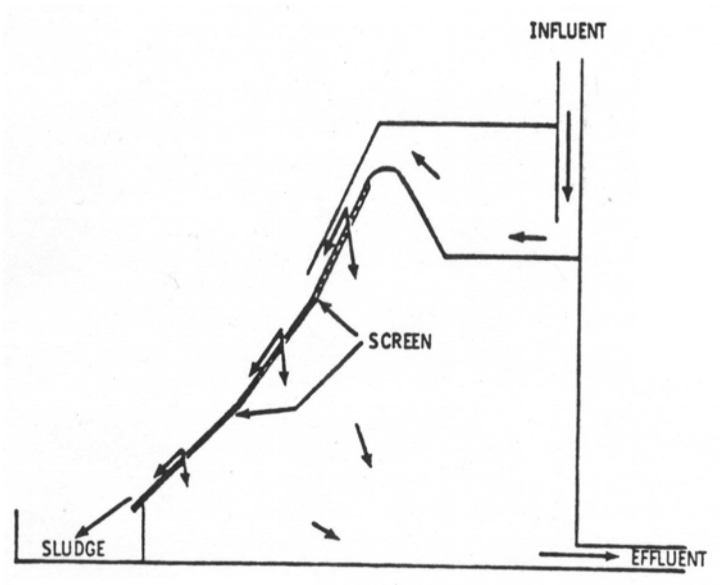

Figure 8-1. General schematic of a static screen (U.S. EPA,1980)

8.2.1.2 Rotary Drum Screens

Rotary drum screens typically are constructed of stainless steel mesh or wedge wire and are designed in one of two ways. The first, driven by external rollers, receives the wastewater at one open end and discharges the solids at the other open end. The screen is inclined toward the exit end to facilitate movement of solids. The liquid passes outward through the screen (usually stainless steel screen cloth or perforated sheet) to a receiver and then to the sewer. To prevent clogging, the screen is usually sprayed continuously from a line of external spray nozzles (USEPA, 1974; USEPA, 1975).

The second type of rotary screen is driven with an external pinion gear. Raw wastewater discharges into the interior of the screen, below the center, and solids are removed in a trough that is mounted lengthwise with a screw conveyor. The liquid exits from the screen into a box where the screen is partially submerged. The screen itself is typically 40 by 40 mesh, with openings of 0.4 mm. To assist lifting the solids to the conveyor trough, perforated lift paddles are mounted lengthwise on the inside surface of the screen. Externally spraying the screen helps reduce blinding, and teflon coated screens reduce clogging by grease. Solid removals up to 82 percent have been reported (USEPA, 1974; USEPA, 1975).

Figure 8-2 shows a general schematic of a rotating drum screen.

8.2.1.3 Brushed Screens

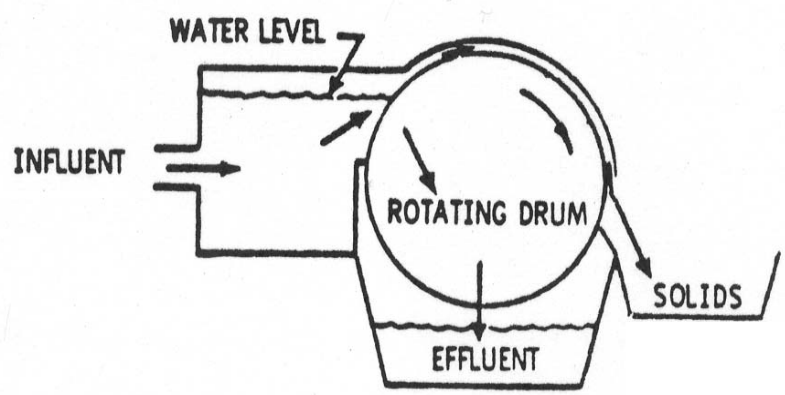

Figure 8-2. General schematic of a rotary drum screen. (U.S. EPA, 1980)

Although most commonly used in sewage treatment, brushed screens can be adapted to remove solids from MPP wastewater. Brushed screens are constructed of a half-circular drum with a stainless steel perforated screen. Mesh size varies according to the type of solid being screened. As influent passes through the screen, rotary brushes sweep across, pushing solids off the screen and into a collection trough. If required, this design can be doubled to dry solid matter further by pushing solids onto a second screen that is pressed and then brushed into the collection trough (Nielsen, 1996).

8.2.1.4 Vibrating Screens

The effectiveness of a vibrating screen depends on a rapid motion. Vibrating screens operate between 99 and 1,800 rpm; the motion can be either circular or straight line, varying from 0.08 to 1.27 cm (1/32 to ½ inch) total travel. Speed and motion are selected by the screen manufacturer for the particular application (USEPA, 1974; USEPA, 1975). Usually made of stainless steel, the vibrating action allows effluent to pass through while propelling solids toward a collection outlet with the aid of gravity (Nielsen, 1996).

Of prime importance in the selection of a proper vibrating screen is the application of the proper cloth. The liquid capacities of vibrating screens are based on the percent of open area of the cloth. The cloth is selected with the proper combination of strength of wire and percent of open area. If the waste solids to be handled are heavy and abrasive, wire of greater thickness should be used to assure long life. However, if the material is light or sticky in nature, the durability of the screening surface may be the least important factor. In such a case, a light wire may be desired to provide an increased percent of open area (USEPA, 1974; USEPA, 1975).

Poultry products facilities may employ two types of vibrating screens. For offal recovery, vibratory screens usually have 20-mesh screening; for feather removal, as well as for in-plant primary treatment of combined wastewater, a 36- by 40-mesh screen cloth is used. On most applications a double-crimped, square-weave cloth is used because of its inherent strength and resistance to wire shifting. Vibratory screens with straight-line action are largely used for byproduct recovery, while those with circular motion are frequently used for in-plant primary treatment (USEPA, 1975).

8.2.2 Catch Basins

Catch basins separate grease and finely suspended solids from wastewater by the process of gravity separation. The basic setup employs a minimum turbulence flowthrough tank where solids heavier than water sink to the bottom, and grease and fine solids rise to the surface. Basins are equipped with a skimmer to remove grease and scum off the top and a scraper to remove sludge at the bottom. The skimmer moves scum into collecting troughs and the scraper moves sludge into a hopper from where both are pumped to byproduct recovery systems. Key factors affecting basin efficiency are detention time and the rate of solid removal from the basin. Depending on influent concentration, recovery rates between 60 and 70 percent can be achieved with a detention time of 20 to 40 minutes (Nielsen, 1996).

Typically, catch basins are rectangular in shape and relatively shallow (1.8 meters or 6 feet is the preferred length). The flow rate is the most important criterion for the design, and the most common sizing factor is determined by measuring the volume of flow during one peak hour with 30 to 40 minutes of detention. An equalization tank before the catch basin reduces size requirements significantly (USEPA, 1974; USEPA, 1975). Depending on the influent characteristics, treatment costs range from 50 to 500 dollars per million gallons treated (FMCITT, 2002).

Tanks can be constructed of concrete or steel; usually two tanks with a common wall are built, in case one becomes unavailable due to maintenance or repairs. Concrete tanks have the inherent advantages of lower overall maintenance and more permanence of structure. However, some facilities prefer to be able to modify their operation for future expansion, alterations, or even relocation. All-steel tanks have the advantage of being semi-portable, more easily field-erected, and more easily modified than concrete tanks. The all-steel tanks, however, require additional maintenance as a result of wear from abrasion and corrosion (USEPA, 1974; USEPA, 1975).

A tank using all-steel walls and a concrete bottom is the best compromise between the all-steel tank and the all-concrete tank. The advantages are the same as for steel; however, the all-steel tank requires a footing underneath and supporting members, whereas the concrete bottom forms the floor and supporting footings for the steel-wall tank (USEPA, 1974; USEPA, 1975).

8.2.3 Dissolved Air Flotation

Dissolved air flotation (DAF) is used extensively in the primary treatment of MPP wastewaters to remove suspended solids. The principal advantage of DAF over gravity settling is its ability to remove very small or light particles (including grease) more completely and in a shorter period of time. Once particles have been to the surface, they are removed by skimming (Metcalf and Eddy, 1991).

In DAF, either the entire influent, some fraction of the influent, or some fraction of the recycled DAF effluent is saturated with air at a pressure of 40 to 50 psi (250 to 300 kPa), and then introduced into the flotation tank (Martin and Martin, 1991). The method of operation may cause operating costs to differ slightly, but process performances are essentially equal among the three modes of operation (USEPA, 1974; USEPA, 1975). With larger wastewater flows, only a fraction of the DAF effluent is saturated and recycled by introduction through a pressure control valve into the influent feed line. From 15 to 120 percent of the influent flow may be recycled in larger units (Metcalf and Eddy, 1991). Under atmospheric pressure in the flotation tank, the air desorbs from solution and forms a cloud of fine bubbles, which transport fine particulate matter to the surface of the liquid in the tank. A skimmer mechanism continually removes the floating solids, and a bottom sludge collector removes any solids that settle. Although unit shape is not important, a more even distribution of air bubbles allows for a shallower flotation tank. Optimum depth settings are between 4 and 9 feet (1.2 to 2.7 meters) (Martin and Martin, 1991).

Chemicals (e.g., polymers and flocculants) are often added prior to the DAF to improve the DAF performance. Typical removals of suspended solids by DAFs vary between 40 to 65 percent without chemical addition and between 80 to 93 percent with chemical addition. Likewise, oil and grease removals by DAF improve from 60 to 80 percent without chemical addition to 85 to 99 percent with chemical addition (Martin and Martin, 1991). There are many advantages to a DAF system, including its low installation costs, compact design, ability to accept variable loading rates, and low level of maintenance (Nielsen, 1996). The mechanical equipment involved in the DAF system is fairly simple, requiring limited maintenance attention for such things as pumps and mechanical drives (USEPA, 1974; USEPA, 1975).

Although alternatives to DAF do exist, including electro flotation, reverse osmosis, and ion exchange, these processes have not been widely adopted by MPP facilities. Cost considerations and technical difficulties associated with these alternatives have prevented ready incorporation of such technologies (Johns, 1995). However, Cowan et al., (1992) summarized treatment and costs for extended trials, using a variety of ultrafiltration and reverse osmosis membranes at a number of slaughterhouses in South Africa. They report that ultrafiltration and reverse osmosis treatment may be the method of choice for treating slaughterhouse wastewaters, both as a pretreatment step prior to discharge to POTW and as a means of reclaiming high quality reusable water from the treated effluent.

8.2.4 Flow Equalization

Since most MPP facilities operate on a five-day per week schedule, weekly variation of wastewater flow is common. In addition, each facility must be thoroughly cleaned and sanitized every 24 hours. Although wastewater flow is relatively constant during processing, a significant difference in flow occurs between processing and cleanup periods, producing a substantial diurnal variation in flow and organic load on days of processing. To avoid the necessity of sizing subsequent treatment units to handle peak flows and loads, in-line flow equalization tanks are installed (Reynolds, 1982; Metcalf and Eddy, 1991). Flow equalization tanks may also be installed to store the effluent from the wastewater treatment plant before discharge to a POTW or other effluent disposal destinations. The end-of-treatment equalization ensures reduced variation in flow and waste load.

Equalization facilities consist of a holding tank and pumping equipment designed to reduce the fluctuations of waste stream. They can be economically advantageous, whether the industry is treating its own wastes or discharging into a city sewer after some pretreatment. The tank is characterized by a varying flow into the tank and a constant flow out. For MPP facilities, flow equalization basins usually are sized to provide a constant 24-hour flow rate on processing days, but also may be sized to provide a constant daily flow rate, including non-processing days. The major advantages of equalization basins are that the subsequent treatment units are smaller, since they can be designed for the 24-hour average flow rather than peak flows, and secondary waste treatment systems operate much better when not subjected to shockloads or variations on feed (USEPA, 1974; USEPA, 1975). To prevent settling of solids and to control odors, aeration and mixing of flow equalization basins are required. Methods of aeration and mixing include diffused air, diffused air with mechanical mixing, and mechanical aeration (Reynolds, 1982; Metcalf and Eddy, 1991).

8.2.5 Chemical Addition

Chemicals are often added to remove pollutants from wastewater. According to the MPP detailed survey responses, chemicals (e.g., polymers, coagulants, and flocculants such as aluminum or iron salts or synthetic organic polymers) are often added to MPP wastewaters prior to DAF or clarifier to aggregate colloidal particles through destabilization by coagulation and flocculation to improve process performance. Essentially all of the chemicals added are removed with the separated solids. When the solids are disposed of by rendering, the use of organic polymers is preferred to avoid high aluminum or iron concentrations in the rendered product produced. EPA noted during site visits to two independent rendering operations that sludges from dissolved air floatation units which use chemical additions to promote solids separation are rendered; however, the chemical bond between the organic matter and the polymers requires that the sludges be processed (rendered) at higher temperatures (260 ^{o}F) and longer retention times. Because the efficacy of aluminum and iron salts and organic polymers is pH dependent, pH adjustment normally precedes the addition of these compounds to minimize chemical use (Ross et. al., 1992; USEPA, 1974; USEPA, 1975).

8.3 SECONDARY BIOLOGICAL TREATMENT

MPP facilities that discharge directly to navigable waters under the authority of a NPDES permit at a minimum apply both primary and secondary treatment to generated wastewaters (see Table 8-1). The objective of secondary treatment is the reduction of BOD through the removal of organic matter, primarily in the form of soluble organic compounds, remaining after primary treatment. Although secondary treatment of wastewater can be performed using a combination of physical and chemical unit processes, use of biological processes has remained the preferred approach (Peavy, 1986). Greater than 90 percent wastewater pollutant removal efficiencies can be achieved with biological treatment (Kiepper, 2001). According to responses to the MPP detailed survey, common systems used for biological treatment of MPP wastewater include lagoons, activated sludge systems, extended aeration, oxidation ditches, and sequencing batch reactors. A sequence of anaerobic biological processes followed by aerobic biological processes is commonly employed by MPP facilities which have biological treatment. Kiepper (2001) suggests that approximately 25 percent of U.S. poultry facilities use biological treatment systems consisting of an anaerobic lagoon followed by an activated sludge system.

8.3.1 Anaerobic Treatment

Anaerobic wastewater treatment processes use the microbially-mediated reduction of complex organic compounds to methane and carbon dioxide as the mechanism for organic matter and BOD reduction. Because methane and carbon dioxide are essentially insoluble in water, both desorb rapidly. This combination of gases, predominantly methane, is commonly referred to as biogas and may be released directly to the atmosphere, collected and flared, or used as a boiler fuel (Clanton, 1997). EPA (1997) provides estimates of the emission factors (e.g., gram-CH_4/head of cattle) for these gases. The BOD removal efficiency by anaerobic treatment can be very high. Anaerobic wastewater treatment processes are more sensitive to temperature and loading rate changes than those of aerobic wastewater treatment processes.

The production of biogas generally occurs as a two-step process. In the first step, complex organic compounds are reduced microbially to simpler compounds, including hydrogen, short-chained volatile acids, alcohols, and carbon dioxide. Carbon dioxide is generated from the reduction of compounds containing oxygen. A wide variety of facultative and anaerobic microorganisms are responsible for these transformations that occur to obtain energy for maintenance, growth, and nutrients, including carbon for cell synthesis (Metcalf and Eddy, 1991; Nielsen, 1996; Peavy, 1986).

In the second step, the short-chained volatile acids, and alcohols are reduced further to methane and carbon dioxide by a group of obligate anaerobic microorganisms referred to collectively as methanogens. This group of microorganisms includes a number of species of methane-forming bacteria with growth rates significantly lower than the facultative and anaerobic microorganisms responsible for the initial reduction of complex compounds into the substrates that are reduced to methane. The biogas produced by the microbial activity typically contains between 30 and 40 percent carbon dioxide and between 60 and 70 percent methane with trace amounts of hydrogen sulfide and other gases (Metcalf and Eddy, 1991; Nielsen, 1996; Peavy, 1986; Clanton 1997).

Due to negligible energy requirements, anaerobic wastewater treatment processes are particularly attractive for the treatment of high strength wastewaters such as MPP wastewaters. Even though anaerobic processes are not capable of producing dischargeable effluents, they can significantly reduce energy requirements for subsequent aerobic treatment to produce dischargeable effluents (Metcalf and Eddy, 1991; Nielsen, 1996; Peavy, 1986; Clanton 1997). Anaerobic treatment can also digest organic solid fractions of animal by-products from slaughterhouse facilities (Banks, 1994; Banks and Wang, 1999).

According to the MPP detailed survey, anaerobic lagoons are the most commonly used anaerobic unit process for treatment of MPP wastewaters. In addition to secondary treatment, anaerobic lagoons provide flow equalization. As noted above, MPP operations normally occur on a 5-day per week schedule, and lagoons reduce variation in daily flows to subsequent secondary and tertiary treatment processes. However, high rate anaerobic processes have continued to attract attention as alternatives to anaerobic lagoons. Included are the anaerobic contact (AC), up-flow anaerobic sludge blanket (UASB), and anaerobic filter processes (AF) (Johns, 1995). These alternatives are especially appealing in situations where land for lagoon construction or expansion is not available.

8.3.1.1 Anaerobic Lagoons

A typical anaerobic lagoon is relatively deep, between 10 and 17 feet (3 to 5 meters) with a detention time of 5 to 10 days. Many treatment systems comprise of at least two lagoons in parallel or series; typical loading rates are between 15 to 20 pounds BOD_5 per 1,000 per cubic feet. The influent wastewater flow is usually near the bottom of the lagoon and has a pH between 7.0 and 8.5. Anaerobic lagoons are not mixed, although some gas mixing occurs. A scum usually develops at the surface, serving several purposes: retarding heat loss, ensuring anaerobic conditions, and reducing emissions of odorous compounds (USEPA, 1974; USEPA, 1975). Depending on the operating conditions, the BOD reductions by anaerobic lagoons can vary widely. Reductions up to 97 percent in BOD_5, up to 95 percent of suspended solids, and up to 96 percent of COD from the influent have been reported (USEPA, 1974; USEPA, 1975, John, 1995).

Wastewater organic carbon anaerobic degradation products emitted from anaerobic lagoons include methane and carbon dioxide. Also, ammonium and hydrogen sulfide are produced from the degradation of sulfur and nitrogen containing compounds found in meat products wastewater. Ammonium can be converted to ammonia in wastewater. The pH of the wastewater determines what emissions are produced in the anaerobic lagoons. A pH of 8 or greater causes more ammonia to be emitted while a pH of 6 or lower produces more hydrogen sulfide and carbon dioxide emissions (Zhang, 2001).

Because odors emitted from anaerobic lagoons can be quite offensive, much effort has been put into maintaining oil and grease caps or developing covers for these ponds. Many operators maintain a cap of oil and grease on the anaerobic lagoons or anaerobic equalization tanks to reduce odors and inhibit oxygen transfer (i.e., promoting anaerobic conditions). This oil and grease cap can be broken up and made ineffective with the influx of storm water or other highly variable flows to the anaerobic lagoons or anaerobic equalization tanks. Synthetic floating or biogas-inflated covers are used to prevent odors from escaping the lagoons, while simultaneously trapping biogas for collection and use as a fuel source. Covering lagoons also reduces heat losses with the result of higher microbial reaction rates. Surface area loading rates can thus be increased and lagoon volume can be reduced (Morris et al., 1998).

8.3.1.2 Alternate Anaerobic Treatment Technologies

Anaerobic Contact Systems

Anaerobic contact systems are very similar to the activated sludge process in concept. Mixed liquor solids from the completely mixed anaerobic reactor vessel are separated in a clarifier and returned to the reactor to maintain a high concentration of biomass (Stebor et al., 1990). The high biomass enables the system to maintains a long solids residence time (SRT) at a relatively short hydraulic retention time (HRT). The completely mixed, sealed reactors are normally heated to maintain a temperature of 35 °C (95 °F).

To provide a relatively short HRT, influent wastewater is mixed with solids removed from the effluent, usually by gravitational settling. Because of the low growth rates of anaerobic microorganisms, as much as 90 percent of the effluent solids may be recycled to maintain an adequate solids residence time. A degasifier that vents methane and carbon dioxide is usually included to minimize floating solids in the separation step (Eckenfelder, 1989). BOD loadings and HRTs range from 2.4 to 3.2 kg/m^3 and from 3 to 12 hours, respectively (USEPA, 1974). Anaerobic contact systems are not common because of high capital cost. Nonetheless, these systems have several advantages over anaerobic lagoons, including the ability to reduce odor problems and reduced land requirements. Biogas produced may be used to maintain reactor temperature.

Up-flow Anaerobic Sludge-Blanket (UASB)

The UASB is another anaerobic wastewater treatment process. Influent wastewater flows upward through a sludge blanket of biologically formed granules, with treatment occurring when the wastewater comes in contact with the granules. The methane and carbon dioxide produced generate internal circulation and serve to maintain the floating sludge blanket. Biogas collection in a gas collection dome occurs above the floating sludge blanket. Particles attached to gas bubbles that rise to the surface of the sludge blanket strike the bottom of degassing baffles, and the degassed particles drop down to the surface of the sludge blanket (Metcalf and Eddy, 1991). Residual solids and granules in the effluent are separated using gravity settling and returned to the sludge blanket. Settling may occur within the reactor or in a separate settling unit. Critical to this operation is the formation and maintenance of granules. Calcium has been used to promote granulation and iron to reduce unwanted filamentous growth (Eckenfelder, 1989).

The application of the UASB process to MPP wastewater has been a less successful endeavor, thus far, compared to other anaerobic processes. For example in treating a slaughterhouse wastewater, it was difficult to generate the sludge granules, thereby significantly lowering the level of BOD removal. High fat concentrations led to the loss of sludge (Johns, 1995).

Anaerobic Filters (AF)

The AF is a column filled with various types of media operating as an attached growth or fixed film reactor. Wastewater flows upward through the column. Because the microbial population is primarily attached to the media, mean cell residence times on the order of 100 days are possible. Thus, it provides an ability to treat very high strength wastewaters with COD concentrations as high as 20,000 mg/L as well as resistance to shock loads. Several studies have shown that AFs operated at short hydraulic retention times can greatly reduce the organic content of process wastewater (Harper et al., 1999). Most development work on the AF has involved high-strength industrial and food-processing wastewaters.

For the MPP industry, removals of COD are reported from 80 to 85 percent when COD loadings are 2 to 3 kg/m^3/day. When loadings are higher, performance suffers. Gas tends to have a relatively high methane content (72 to 85 percent). One facility reported BOD concentrations below 500 mg/L, at 33°C, with a COD loading of 2 to 3 kg/m^3/day. It is important to have effective pretreatment to remove oil and grease and suspended solids, as a high oil and grease concentration can cause unstable operation of the system (Harper et al., 1999; Johns, 1995).

Based on pilot-scale experiments, anaerobic-packed bed treatment has proven to be an effective alternative to DAF for pretreatment of poultry processing wastewater (Harper et al., 1999).

Anaerobic Sequence Batch Reactor (ASBR)

The ASBR is a variation of the anaerobic contact process that eliminates the need for complete mixing. This treatment is particularly applicable for MPP wastewaters, because high protein concentrations eliminate the need for supplemental alkalinity. In addition, ASBR easily addresses high levels of solids that are typically found in MPP wastewaters. One study that used an ASBR system on process wastewater achieved BOD_5 removals ranging from 37 to 77 percent and COD removals ranging from 27 to 63 percent. The resulting biogas was 73 to 81 percent methane, although the high concentration of hydrogen sulfide (~1,800 ppm) in the biogas may make at least partial removal of hydrogen sulfide prior to use as a fuel (Morris et al., 1998).

8.3.2 Aerobic Treatment

In the treatment of MPP wastewaters, aerobic treatment may directly follow primary treatment, or more typically follow some form of anaerobic treatment to reduce BOD and suspended solids concentrations to levels required for discharge. Reduction of ammonia also is a typical role of aerobic processes in the treatment of MPP wastewaters. Many NPDES permits are written with seasonal limits for ammonia, because the lower pH and lower temperature of the receiving waters during winter reduce the toxicity of ammonia by converting it to ammonium (Ohio EPA, 1999). Advantages of using aerobic wastewater treatment processes include low odor production, fast biological growth rate, no elevated operation temperature requirements; and quick adjustments to temperature and loading rate changes. However, the operating costs of aerobic systems are higher than the costs of anaerobic systems for processing livestock wastewater, because of the relatively high space, maintenance, management, and energy required for artificial oxygenation. The microorganisms involved in aerobic treatment process require free dissolved oxygen to reduce the biomass in the wastewater (Clanton, 1997).

Aerobic wastewater treatment processes can be broadly divided into suspended and attached growth processes. Aerobic lagoons and various forms of activated sludge process like conventional, extended aeration, oxidation ditches, and sequencing batch reactors (SBRs) are examples of suspended growth processes; trickling filters and rotating biological contactors (RBCs) are examples of attached growth processes. Both utilize a diverse population of heterotrophic microorganisms using molecular oxygen in the process of obtaining energy for cell maintenance and growth (Metcalf and Eddy, 1991).

Aerobic wastewater treatment processes have the primary objective of transforming soluble and colloidal organic compounds into microbial biomass, with subsequent removal of the biomass formed by settling or mechanical separation as the primary mechanism for organic matter and BOD removal. Some oxidation of organic carbon to carbon dioxide also occurs to provide energy for cell maintenance and growth. The degree of carbon oxidation depends on the solids retention time (SRT), also referred to as the mean cell residence time of the process, which determines the age of the microbial population. Processes with long SRTs operate in the endogenous respiration phase of the microbial growth curve and generate less settleable solids per unit BOD removed. Attached growth processes generally operate at long SRTs (Metcalf and Eddy, 1991).

At SRTs sufficiently long to maintain an active population of nitrifying bacteria, oxidation of ammonia nitrogen to nitrate nitrogen (nitrification) also occurs. However, the rates of growth of the autotrophic bacteria responsible for nitrification, *Nitrosomas* and *Nitrobacter*, are substantially slower than the growth rates of the microorganisms responsible for BOD reduction (Metcalf and Eddy, 1991). Therefore, the amount of nitrification during aerobic treatment will depend on the type of treatment system used and its operating conditions.

8.3.2.1 Activated Sludge

The activated sludge process (see Figure 8-3) is one of the most commonly used biological wastewater treatment processes in the United States (Metcalf and Eddy, 1991). According to the MPP detailed survey, various forms of activated sludge process used in the

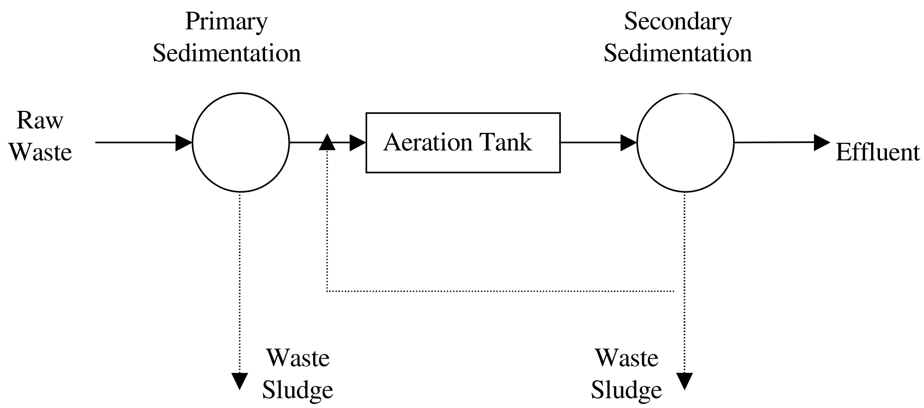

Figure 8-3. Activated Sludge Process (USEPA, 1974).

MPP industry include conventional, complete mix, extended aeration, oxidation ditch, and sequencing batch reactor. Other forms of activated sludge include tapered aeration, step-feed aeration, modified aeration, contact stabilization, Kraus process, and high-purity oxygen. All of these forms share the common characteristics of short HRTs, usually no more than several hours, and SRTs on the order of 5 to 15 days. This differential is maintained by continually recycling a fraction of the settleable solids separated after aeration by clarification back to the aeration basin. These settled solids contain an active, adapted microbial population and are the source of the term "activated sludge." The microbial population is comprised primarily of bacteria and protozoa, which aggregate to form flocs.

Floc formation is a critical factor in determining the efficacy of settling after aeration, which is the primary mechanism of BOD and suspended solids reduction. The fraction of activated sludge returned, known as the recycle ratio, determines the SRT of the process and serves the basis for controlling process performance. Typically, about 20 percent of the settled solids are recycled to maintain the desired concentration of mixed liquor suspended solids (MLSS). The remaining sludge is removed from the system and may be stabilized using aerobic or anaerobic digestion or by chemical addition (lime stabilization), which may be followed by dewatering by filtration or centrifugation (USEPA, 1974; USEPA, 1975).

The activated sludge process is capable of 95 percent reductions in BOD_5 and suspended solids (USEPA, 1974; USEPA, 1975). In addition, reductions in ammonia nitrogen in excess of

95 percent are possible at temperatures above 10°C and dissolved oxygen concentrations above 2 mg/L (Johns, 1995). Performance depends on maintaining an adequate SRT and mixed liquor suspended solids with good settling characteristics, which depends on floc formation. Excessive growth of filamentous organisms can impair activated sludge settleability. Excessive mixing can lead to the formation of pin flocs, which also have poor settling characteristics. Diffused air used for achieving the required aeration and mechanical systems used for obtaining necessary mixing result in significant energy use (Metcalf and Eddy, 1991).

Conventional

In the conventional activated sludge process, the aeration tank is a plug flow reactor. Plug flow regime may be made with baffles in aeration tanks. Settled wastewater and recycled activated sludge enter the head end of the aeration tank and are mixed by diffused-air or mechanical aeration. Air application is generally uniform throughout tank length. During the aeration period, adsorption, flocculation, and oxidation of organic matter occurs. Activated-sludge solids are separated in a secondary settling tank (Metcalf and Eddy, 1991).

Complete-mix

Complete mix activated sludge process uses a complete mix tank as an aeration basin. The process is an application of the flow regime of a continuous-flow stirred tank reactor. Settled wastewater and recycled activated sludge are introduced typically at several at several points in the aeration tank. The organic load on the aeration tank and the oxygen demand are uniform throughout the tank length (Metcalf and Eddy, 1991).

Extended Aeration

Extended aeration is another variant of the activated sludge process. The principal difference between extended aeration and the other variants of the activated sludge process is that extended aeration operates in the endogenous respiration phase of the microbial growth curve. Thus, lower organic loading rates and longer HRTs are required. Because of longer HRTs, typically 18 to 36 hours, extended aeration has the ability to absorb shock loads. Other advantages include its generation of less excess solids from endogenous respiration and greater overall process stability (USEPA, 1974). However, poor settling characteristics of aeration basin effluent is a frequently encountered problem with extended aeration. Generally, extended aeration treatment facilities are prefabricated package unit operations used for treating relatively low volume wastewater flows for small communities (Metcalf and Eddy, 1991). Extended aeration can be designed to provide high degree of nitrification.

Oxidation Ditches

An oxidation ditch system represents a modification of the activated sludge process in terms of its reactor configuration. The oxidation ditch consists of a ring- or oval-shaped channel and is equipped with mechanical aeration devices (Metcalf and Eddy, 1991). Aerators in the form of brush rotors, disc aerators, surface aerators, draft tune aerators, or fine pore diffusers with submersible pumps provide oxygen transfer, mixing and circulation in the oxidation ditch. Wastewater enters the ditch, is aerated, and circulates at about 0.8 to 1.2 ft/s. Oxidation ditches typically operate in an extended aeration mode with HRT greater than 10 hours and SRT of 10 to 50 days (USEPA, 1993). Oxidation ditches provide high removal of BOD and can be designed for nitrification and nitrogen and phosphorous removal (Sen et al., 1990).

Sequencing Batch Reactor

The sequencing batch reactor (SBR) is a fill-and-draw type reactor system using one or more complete mix tanks in which all steps of activated sludge process occur. SBR systems have four basic periods: Fill (the receiving of raw wastewater), React (the time to complete desired reaction), Settle (the time to separate the microorganisms from treated effluent), and Idle (the time after discharging the tank and before refilling). However, these periods may be modified or eliminated depending on effluent requirements. The time for a complete cycle is the total time between the beginning of Fell and the end of Idle (Martin and Martin, 1991). SBR systems provides high removal of BOD and suspended solids. In addition, SBR systems can be designed for nitrification and to remove nitrogen and phosphorous. Lo and Liao (1990) report that SBR technology can be used successfully in the treatment of poultry processing wastewaters for the removal of BOD^5 and nitrogen. SBR offers the advantages of operational and loading flexibility, high removal efficiency, competitive capital costs, and reduced operator maintenance (Glenn et al., 1990).

8.3.2.2 Lagoons

Lagoons are widely used in the treatment of MPP wastewater. They are comparatively cheaper than other treatment processes, although they require larger land area. Lagoons can be anaerobic, aerobic, aerated, or facultative. Anaerobic lagoons are discussed in Section 8.3.1.1. Other types of lagoons are discussed in this section.

Aerobic lagoons

Aerobic lagoons, which are also known as aerobic stabilization ponds, are large shallow earthen basins that use algae in combination with other microorganisms for wastewater treatment. Low-rate ponds, which are designed to maintain aerobic conditions throughout the liquid column, may be up to five feet deep. High-rate ponds are usually shallower, with a maximum depth of no greater than 1.5 feet. They are designed to optimize the production of algal biomass as a mechanism for nutrient removal. In aerobic stabilization ponds, oxygen is supplied by a combination of natural surface aeration and photosynthesis. In the symbiotic relationship between the algae and other microorganisms present, the oxygen released by the algae during photosynthesis is used by the non photosynthetic microorganisms present in the aerobic degradation of organic matter, while the nutrients and carbon dioxide released by the nonphotosynthetic microorganisms are used by the algae (Martin and Martin, 1991).

Loading rates of aerobic stabilization ponds are in the range of 10 to 300 pounds of BOD per acre per day with an HRT of 3 to 10 days. Soluble BOD_5 reductions of up to 95 percent are possible with aerobic stabilization ponds (Martin and Martin, 1991). Aerobic stabilization ponds may be operated in parallel or series. To maximize performance, intermittent mixing is necessary. Without supplemental aeration, dissolved oxygen concentrations will vary from super saturation due to photosynthesis during day light hours, to values at or approaching zero at night, especially with high-rate ponds. Also, settled solids will create an anaerobic zone at the bottom of the pond (Reynolds, 1982). Thus, nitrogen removal is achieved by the combined processes of nitrification and denitrification.

The low cost of aerobic stabilization ponds is offset, especially in colder climates, by seasonal variation in performance. In winter, limited sunlight due to shorter day length and cloud cover limits photosynthetic activity and oxygen release, as well as algae growth. In addition, ice cover limits natural surface aeration. Thus, aerobic stabilization ponds in colder climates may become anaerobic lagoons in winter months with a concurrent deterioration in effluent quality and a source of noxious odors in the following spring before predominately aerobic conditions become reestablished (Martin and Martin, 1991). Scaief (1975), however, reports no difference in overall treatment efficiency across all seasons for anaerobic-aerobic lagoon systems or anaerobic contact process followed by aerobic lagoons.

Aerated Lagoons

Aerated lagoons are earthen basins used in place of concrete or steel tanks for suspended growth biological treatment of wastewater. Aerated lagoons typically are about 8 feet (2.4 m) deep, but can be as much as 15 feet (4.6 m) deep and may be lined to prevent seepage of wastewater to ground water. Although diffused air systems are used for aeration and mixing, fixed and floating mechanical aerators are more common.

Natural aeration occurs in diffused air systems by air diffusion at the water surface by wind- or thermal-induced mixing and by photosynthesis. Algae and cyanobacteria (blue-green algae) are the microorganisms responsible most of the photosynthetic activity in a naturally-aerated lagoon. Naturally aerated lagoons are approximately 1 to 2 feet deep, so that sunlight can penetrate the full lagoon depth to maintain photosynthetic activity throughout the day. Mechanically aerated lagoons do not have a depth requirement, because oxygen is supplied artificially instead of by algal photosynthesis (Zhang, 2001).

Aerated lagoons can be operated as activated sludge units with the recycle of settled solids with relatively short HRTs, or as complete mix systems without settled solids recycle. Systems operated as activated sludge units have a conventional clarifier for recovery of settled solids for recycle. Aerated lagoons operated as complete mix systems without solids recycle may use a large, shallow earthen basin in place of a more conventional clarifier for removal of suspended solids. Typically, these basins also are used for the storage and stabilization of the settled solids. Usually a detention time of no less than 6 to 12 hours is required.

One of the principal advantages of aerated lagoons is relatively low capital cost. However, more land is required. With earthen settling basins, algae growth and odors can be problems, along with consistent effluent quality.

Facultative Lagoons

The facultative lagoons are deeper than aerobic lagoons, varying in depth from 5 to 8 ft. Waste is treated by bacterial action occurring in an upper aerobic layer, a facultative middle layer, and a lower anaerobic layer. Aerobic bacteria degrade the waste in the upper layer, where oxygen is provided by natural surface aeration and algal photosynthesis. Settleable solids are deposited on the lagoon bottom and degraded by anaerobic bacteria. The facultative bacteria in the middle layer degrade the waste aerobically, whenever dissolved oxygen is present and anaerobically otherwise. The facultative lagoons have more depth and smaller surface areas aerated or aerobic lagoons but still have good odor control capabilities, because of the presence of the upper aerobic layer, where odorous compounds such as sulfides produced by anaerobic degradation in the lower layer, are oxidized before emission into the atmosphere. Biochemical reactions in the facultative lagoons are a combination of aerobic and anaerobic degradation reactions (Zhang, 2001).

8.3.2.3 Alternate Aerobic Treatment Technologies

Trickling Filters

A trickling filter consists of a bed of highly permeable media to which a microbial flora becomes attached, a distribution system to spread wastewater uniformly over the bed surface, and an under-drain system for collection of the treated wastewater and any microbial solids that have become detached from the media. As the wastewater percolates or trickles down through the media bed, the organic material present is absorbed onto the film or slime layer of attached microorganisms. Within 0.1 to 0.2 mm of the surface of the slime layer, the organic matter absorbed is metabolized aerobically, providing energy and nutrients for cell maintenance and growth. As cell growth occurs, the thickness of the slime layer increases and oxygen diffusing into the slime layer is consumed before penetration to the media surface occurs and anaerobic conditions develop near the media surface. In addition, organic matter and nutrients necessary for cell maintenance and growth are lacking due to utilization near the surface of the slime layer. Thus, endogenous conditions develop near the media surface and detachment occurs from hydraulic shear forces as the microorganisms at and near the media surface die. This process is known as "sloughing" and may be a periodic or continual process depending on organic and hydraulic loading rates. Hydraulic loading rate usually is adjusted to maintain continual sloughing and a constant slime layer thickness (Metcalf and Eddy, 1991).

The biological community in the trickling filter process includes aerobic, facultative, and anaerobic bacteria, fungi, and protozoans. The aerobic microbial population may include the nitrifying bacteria *Nitrosomonas* and *Nitrobacter*. It also may include algae and higher organisms such as worms, insect larvae, and snails, unlike activated sludge processes. Variations in these biological communities occur according to individual filter and operating conditions (Metcalf and Eddy, 1991).

Trickling filters have been classified as low-rate, intermediate-rate, high-rate, super high-rate, roughing, and two-stage, based on filter medium, hydraulic and BOD_5 loading rates, recirculation ratio, and depth (Metcalf and Eddy, 1991). Hydraulic loading rates range from 0.02 to 0.06 gallon per ft^2-day for low-rate filters to 0.8 to 3.2 gallon per ft^2-day for roughing filters. Organic loading rates range from 5 to 25 pounds BOD_5 per 10^3 ft^2-day to 100 to 500 pounds BOD_5 per 10^3 ft^2-day. Both low-rate and two-stage trickling filters can produce a nitrified effluent while roughing filters provide no nitrification. Others may provide some degree of nitrification. Low-rate and intermediate-rate trickling filters traditionally have used rock or blast furnace slag as filter media while high-rate filters only employ rock. Super high-rate filters use plastic media, while roughing filters may be constructed using either plastic or redwood media; two-stage filters may use plastic or rock media (Metcalf and Eddy, 1991).

Trickling filters are secondary wastewater treatment unit processes and require primary treatment for removal of settleable solids and oil and grease to reduce the organic load and prevent plugging. Secondary clarification also is necessary. Lower energy requirements make trickling filters attractive alternatives to activated sludge processes. However, mass-transfer limitations limit the ability of trickling filters to treat high strength wastewaters. To successfully treat such wastewaters, a two- or three-stage system is necessary. When staging of filters is used, a clarifier usually follows each stage. The overall BOD_5 removal efficiency of can be as great as 95 percent (USEPA, 1974).

Rotating Biological Contactors

Rotating biological contactors (RBCs) also employ an attached film or slime layer of microorganisms to adsorb and metabolize wastewater organic matter, providing energy and nutrients for cell maintenance and growth. RBCs consist of a series of closely spaced circular disks of polystyrene or polyvinyl chloride mounted on a longitudinal shaft. The disks are rotated alternately, exposing the attached microbial mass to the wastewater being treated for adsorption of organic matter and nutrients and then the atmosphere for adsorption of oxygen. The rate of rotation controls oxygen diffusion into the attached microbial film and provides the sheer force necessary for continual biomass sloughing (Metcalf and Eddy, 1991). Mass transfer limitations limit the ability of RBCs to treat high strength wastewaters, such as MPP wastewaters. RBCs can be operated in series like multi-stage trickling filter systems, a tapered feed arrangement is possible. An example of such an arrangement would be three RBCs in parallel in stage one, followed by two RBCs in parallel in stage two, and one RBC in stage three.

As with trickling filters, hydraulic and organic loading rates are criteria used for design. Design values may be derived from pilot plant or full-scale performance evaluations or using the theoretical or empirical approaches (Metcalf and Eddy, 1991). Typical hydraulic and organic loading rate design values for secondary treatment are 2 to 4 gallon/ft^2-day and 2.0 to 3.5 pounds total $BOD_5/10^3$ ft^2-day, respectively with effluent BOD_5 concentrations ranging from 15 to 30 mg/L. For secondary treatment combined with nitrification, typical hydraulic and organic loading rate design values for are 0.75 to 2 gallon/ft^2-day and 1.5 to 3.0 pounds $BOD_5/10^3$ ft^2-day, respectively producing effluent BOD_5 concentrations between 7 and 15 mg/L and NH_3 concentrations of less than 2 mg/L (Metcalf and Eddy, 1991).

The major advantages of RBCs are: (1) relatively low installation cost, (2) ability to combine secondary treatment with ammonia removal by nitrification, especially in multi-stage systems, and (3) resistance to shock loads. The major disadvantage is the need to enclose them, especially in cold climates to maintain high removal efficiencies, control odors, and minimize problems with temperature sensitivities (USEPA, 1974). Early RBC units experienced operating problems, including shaft and bearing failures, disk breakage, and odors. Design modifications have been made to address these problems, including increased submergence to reduce shaft and bearing loads (Metcalf and Eddy, 1991).

Although RBCs are used in both the United States and Canada for secondary treatment of domestic wastewaters, use for secondary treatment of high strength industrial wastewaters such as MPP wastewaters has been limited. Energy requirements associated with activated sludge processes may make RBCs more attractive for treating MPP wastewaters, especially following physical/chemical and anaerobic pretreatment. A BOD_5 reduction of 98 percent is achievable with a four-stage RBC (USEPA, 1974).

8.4 TERTIARY TREATMENT

Tertiary or advanced wastewater treatment generally is considered to be any treatment beyond conventional secondary treatment to remove suspended or dissolved substances. Tertiary wastewater treatment can have one or several objectives. One common objective is further reduction in suspended solids concentration after secondary clarification. Nitrogen and phosphorus removal also are common tertiary wastewaters treatment objectives. Existing wastewater treatment plants may be retrofit without the addition of new tanks or lagoons to incorporate biological nutrient removal (Randall et al., 1999). In addition, tertiary wastewater treatment may be used to remove soluble refractory, toxic, and dissolved inorganic substances. In the treatment of MPP wastewaters, tertiary wastewater treatment most commonly is used for further reductions in nutrients and suspended solids.

8.4.1 Nutrient Removal

In primary and secondary wastewater treatment processes, some reduction of nitrogen and phosphorus occurs by the separation of particulate matter during settling or cell synthesis. However, limited assimilative capacity of receiving waters may require additional reductions in nitrogen and phosphorus concentrations before discharge. Both biological and physicochemical unit processes can be used to reduce nitrogen and phosphorous concentration in wastewater. Biological processes are generally more cost effective than physicochemical processes. Moreover, retrofit existing secondary treatment systems for biological nutrient removal may lead to reduced costs given their lower requirements for energy use and chemical addition (Randall and Mitta, 1998; Randall et al., 1999).

8.4.1.1 Nitrogen Removal

The removal of nitrogen from wastewaters biologically is a two-step process, beginning with nitrification and followed by denitrification. Nitrification, a microbially-mediated process, also is a two-step process, beginning with the oxidation of ammonia to nitrite and followed by the oxidation of nitrite to nitrate. Bacteria of the genus *Nitrosomonas* are responsible for the oxidation of ammonia to nitrite; bacteria of the genus *Nitrobacter* are responsible for the subsequent oxidation of nitrite to nitrate (Metcalf and Eddy, 1991).

Following the nitrification process under anaerobic conditions, nitrite and nitrate are reduced microbially by denitrification producing nitrogen gas as the principal end product. Small amounts of nitrous oxide and nitric oxide also may be produced, depending on environmental conditions. Because nitrogen, nitrous oxide, and nitric oxide are essentially insoluble in water, desorption occurs immediately. Although nitrification can occur in combination with secondary biological treatment, denitrification generally is a separate unit process following secondary clarification. Because the facultative and anaerobic microorganisms responsible for denitrification are heterotrophs, denitrification after secondary clarification requires the addition of a source of organic carbon for cell maintenance and growth. Methanol probably is the most commonly added source of organic carbon for denitrification, although raw wastewater (by-passed to the denitrification treatment tank), biosolids, and a variety of other substances also can be used (USEPA, 1993, Metcalf and Eddy, 1991).

The chemical transformations that occur during nitrification and denitrification are outlined below (Metcalf and Eddy, 1991):

Nitrification*:*

$NH_4^+ + 1.5\ O_2 \longrightarrow NO_2^- + 2H^+ + H_2O$ (*Nitrosomonas*)

$NO_2- + 0.5\ O_2 \longrightarrow 2NO_3^-$ (*Nitrobacter*)

Denitrification (using methanol as carbon source):

$$NO_3^- + 1.08\ CH_3OH + H^+ \longrightarrow 0.065\ C_5H_7O_2N + 0.47\ N_2 + 0.76\ CO_2 + 3 + 2.44\ H_2O$$

Nitrification unit processes can be classified based on the degree of separation of the oxidation of carbonaceous and nitrogenous compounds respectively to carbon dioxide and nitrate (Metcalf and Eddy, 1991). Combined carbon oxidation and nitrification can be achieved in all suspended growth secondary wastewater treatment processes and with all attached growth processes except roughing filters. Carbon oxidation and nitrification processes may also be separated, with carbon oxidation occurring first, using both suspended and attached growth processes in a variety of combinations. Both suspended and attached growth processes are used for denitrification, following combined carbon oxidation and nitrification.

Nitrification and denitrification can be combined in a single process. With this approach, wastewater organic matter serves as the source of organic carbon for denitrification. Thus, the cost of adding a supplemental source of organic carbon and providing re-aeration after denitrification is eliminated. Also eliminated is the need for intermediate clarifiers and return sludge systems. The proprietary four-stage Bardenpho process (Metcalf and Eddy, 1991) is a combined nitrification-denitrification process using both organic carbon in untreated wastewater and organic carbon released during endogenous respiration for denitrification. Separate aerobic and anoxic zones provide for nitrification and then denitrification.

Other processes include the Modified Ludzack-Ettinger (MLE) process, A^2/O, University of Capetown (UCT) (USEPA, 1993). The A^2/O, and University of Capetown (UCT) process was developed to remove both nitrogen and phosphorous. Sequencing batch reactors (SBR) can also be used to achieve nitrification and denitrification (USEPA, 1993). Biological nitrogen and phosphorus removals can be enhanced in oxidation ditch systems by controlling aeration to maintain reliable aerobic, anoxic, and anaerobic volumes. For example, a BNR oxidation ditch process developed by Virginia Tech for retro-fitting a domestic wastewater treatment facility was capable of: (1) maintaining less than 0.5 mg/L total phosphorus and between 3 and 4 mg/L for total nitrogen in the discharged effluent all year round and (2) significantly reducing operational costs by reducing electrical energy, aeration, and chemical addition (Sen et al., 1990).

Nitrification is easily inhibited by a number of factors including toxic organic and inorganic compounds, pH, and temperature. In poorly buffered systems, the hydrogen ions released when ammonia is oxidized to nitrite/nitrate can reduce pH to an inhibitory level without the addition of a buffering agent.

A pH of at least 7.2 is generally recognized as necessary to maintain a maximum rate of nitrification (Grady and Lim, 1980). Based on the following theoretical stoichiometric relationships for the growth of *Nitrosomonas* and *Nitrobacter,* the alkalinity (HCO_3^-) utilized is 8.64 mg HCO_3^- per mg of ammonia nitrogen oxidized to nitrate nitrogen. For *Nitrosomonas*, the equation is:

$$55\ NH_4^+ + 76\ O_2 + 109\ HCO_3^- \rightarrow C_5H_7O_2N + 54\ NO_2^- + 57\ H_2O + 104\ H_2CO_3$$

For *Nitrobacter*, the equation is:

$$400\ NO_2^- + NH_4^+ + 4\ H_2CO_3 + HCO_3^- + 195\ O_2 \rightarrow C_5H_7O_2N + 3\ H_2O + 400\ NO_3^-$$

As noted above, one of the advantages of using wastewater organic matter as the source of organic carbon for denitrification is the elimination of the cost of an organic carbon source such as methanol. A second advantage is elimination of the need to add a source of bicarbonate alkalinity in poorly buffered systems to compensate for the utilization of alkalinity resulting from nitrification and the associated reduction in pH. As shown in the overall energy reaction for nitrification, two hydrogen ions are released for every ammonium ion oxidized to nitrate.

$$NH_4^+ + 2\ O_2 \rightarrow NO_3^-\ 2\ H^+\ H_2O$$

However, denitrification releases one hydroxyl ion for each nitrate ion reduced to nitrogen gas, as shown in the following overall energy reaction for denitrification using methanol as the source of organic carbon.

$$6\ NO_3^- + 5\ CH_3OH \rightarrow 5\ CO_2 + 3\ N_2 + 7\ H_2O + 6\ OH^-$$

In addition, hydrogen ions are required for cell synthesis during denitrification, as shown by the following relationship:

$$3\ NO_3^- + 14\ CH_3OH + CO_2 + 3\ H^+ \rightarrow 3\ C_5H_7O_2N + H_2O$$

Therefore, using wastewater organic matter as the source of organic carbon for denitrification in a combined nitrification denitrification system generally eliminates the need for adding a source of alkalinity to prevent pH inhibition of nitrification. Very poorly buffered systems are the exception.

Using wastewater organic matter as the source of organic carbon for denitrification also reduces aeration requirements for BOD removal in suspended growth systems. Based on half reactions for electron acceptors, 1/5 mole of NO_3^- is equivalent to 1/4 mole of O_2. Therefore, each unit mass of NO_3^- - N is equivalent to 2.86 units of O_2 in its ability to oxidize organic matter, if cell synthesis is ignored. However, some organic matter must be converted into cellular material and is not completely oxidized. It does, however, represent the removal of BOD through removal of excess suspended solids and an additional reduction in aeration requirements for BOD removal. Therefore, the actual reduction in BOD realized by using wastewater organic matter as the source of organic carbon for denitrification is marginally higher that 2.86 mass units of BOD per unit NO_3^- - N denitrified. The magnitude of this marginal increase depends on the SRT in the denitrification reactor with the magnitude decreasing as SRT increases. Assuming a

SRT of 7.5 days, a ratio of BOD_5 in wastewater used as an organic carbon source for denitrification to NO_3^- - N of 3.5 should provide for essentially complete denitrification.

An added positive consequence of using wastewater organic matter as the source of organic carbon for denitrification is that sludge production per unit BOD removed is lower, because denitrification is an anoxic process occurring under anaerobic conditions. Typical cell yield under anaerobic conditions is 0.05 mg volatile suspended solids (VSS) per mg BOD removed versus 0.6 mg VSS per mg BOD removed under aerobic conditions (Metcalf and Eddy, 1991).

Both *Nitrosomonas* and *Nitrobacter* are autotrophic mesophilic microorganisms with relatively low growth rates in comparison to heterotrophs, even under optimal conditions. Thus, maintaining an actively nitrifying microbial population may become harder and require excessively long SRTs in cold weather (Metcalf and Eddy, 1991; USEPA, 1993).

8.4.1.2 Phosphorus Removal

To achieve low effluent discharge limits, phosphorous may be removed from wastewater biologically and/or by physicochemical methods. Biological treatment is cheaper than physicochemical methods and is particularly suitable for facilities with high flows.

Biological Treatment

Microorganisms used in secondary wastewater treatment require phosphorus for cell synthesis and energy transport. In the treatment of typical domestic wastewater, between 10 and 30 percent of influent phosphorus is removed by microbial assimilation, followed by clarification or filtration. However, phosphorus assimilation in excess of requirements for cell maintenance and growth, known as luxury uptake, can be induced by a sequence of anaerobic and aerobic conditions (Metcalf and Eddy, 1991).

Acinetobacter is one of the organisms primarily responsible for the luxury uptake of phosphorus in wastewater treatment. In response to volatile fatty acids present under anaerobic conditions, stored phosphorus is released. However, luxury uptake and storage for subsequent use of phosphorus occurs when anaerobic conditions are followed by aerobic conditions. Thus, removal of phosphorus by clarification or filtration following secondary treatment is increased, because biosolids are already wasted (USEPA, 1987, Metcalf and Eddy, 1991; Reddy, 1998).

Currently, several proprietary processes use luxury uptake for removal of phosphorus from wastewater during suspended growth secondary treatment. Included are the A/O, PhoStrip, and Bardenpho processes. In addition, sequencing batch reactors (SBRs) can be operated to remove phosphorus. In the PhoStrip process, phosphorus is stripped from the biosolids generated using anaerobic conditions to stimulate release. The soluble phosphorus generated then is precipitated using lime. Both the A/O and PhoStrip processes are capable of producing final effluent total phosphorus concentrations of less than 2 mg/L. A modified version of the A/O process, the A^2/O process, along with the Bardnepho process and SBRs are capable of combined biological removal of nitrogen and phosphorus (USEPA, 1987; Metcalf and Eddy, 1991; Reddy, 1998).

Physicochemical Process

Phosphorus can be removed from wastewater by precipitation using metal salts or lime. The metal salts most commonly used are aluminum sulfate (alum) and ferric chloride. However, ferrous sulfate and ferrous chloride also can be used. Use of lime is less common due to operating and maintenance problems associated with its use and the large volume of sludge produced. Polymers often are used in conjunction with metal salts to improve the degree of phosphorus removal. Ion exchange, discussed in Section 8.4.3.3, also is an option for phosphate phosphorus removal, but is rarely used in wastewater treatment. (Metcalf and Eddy, 1991).

Chemicals can be added to remove phosphorus in: (1) raw wastewater prior to primary settling, (2) primary clarifier effluent, (3) mixed liquor with suspended growth treatment processes, (4) effluent from biological treatment processes prior to secondary clarification, or (5) after secondary clarification (Metcalf and Eddy, 1991). In Option 1 (pre-precipitation), precipitated phosphorus is removed with primary clarifier solids, whereas removal is with secondary clarifier solids for Options 2 through 4 (co-precipitation). In Option 5, additional clarification or filtering facilities are required. In the treatment of MPP wastewaters, the addition of chemicals for phosphorus removal prior to dissolved air flotation is a possible option (Metcalf and Eddy, 1991).

With alum addition, phosphorus is precipitated as aluminum phosphate ($AlPO_4$), and aluminum hydroxide ($Al(OH)_3$). With the addition of ferric chloride, the chemical species produced are ferric phosphate ($FePO_4$) and ferric hydroxide ($Fe[OH]_3$). Lime addition produces calcium phosphate ($Ca_5[PO_4]_3[OH]$), magnesium hydroxide ($Mg[OH]_2$), and calcium carbonate ($CaCO_3$). In the case of alum and iron, one mole theoretically will precipitate one mole of phosphate. However, competing reactions and the effects alkalinity, pH, trace elements, and ligands found in wastewater make bench-scale or full-scale tests necessary to determine dosage rates. Due to coagulation and flocculation, removal of suspended solids also occurs with the precipitated phosphorus species. With the addition of aluminum and iron salts, the addition of a base to maintain a pH in the range of 5 to 7 to optimize the efficacy of phosphorus precipitation may be necessary depending on wastewater buffer capacity. (USEPA, 1987; Metcalf and Eddy, 1991; Reddy, 1998).

When lime is used, it usually is calcium hydroxide ($Ca(OH)_2$). Due to reaction with natural bicarbonate alkalinity forming $CaCO_3$ as a precipitate, an increase to a pH of 10 or higher is necessary for the formation of $Ca_5(PO_4)_3(OH)$. After lime is used to precipitate phosphorus, recarbonation with carbon dioxide is necessary to lower pH (USEPA, 1987; Metcalf and Eddy, 1991; Reddy, 1998).

When chemical addition is used for phosphorus removal, additional benefits are realized. Due to coagulation and flocculation, effluent BOD and suspended solids concentrations also are reduced, especially when chemical addition occurs after secondary clarification (USEPA, 1987; Metcalf and Eddy, 1991; Reddy, 1998).

8.4.2 Residual Suspended Solids Removal

Simple clarification after secondary wastewater treatment may not reduce the concentration of suspended solids to the level necessary to comply with concentration or mass discharge permit limits or both. Granular-medium filtration usually is used to achieve further reductions in suspended solids concentrations. This practice also provides further reductions in BOD. Filtration is a solid-liquid separation in which the liquid passes through a porous material to remove as much fine material as possible (Reynolds, 1982).

Granular Medium Filters

Metcalf and Eddy (1991) lists nine different types of commonly used granular-medium filters. They are classified as either semi-continuous or continuous, depending on whether back washing is a batch or a semi continuous or continuous operation. Within each classification, there are several different types, depending on bed depth, type of filtering medium, and stratification or lack thereof of the filtering medium. Shallow, conventional, and deep bed filters respectively are typically about 11 to 16, 30 to 36, and 72 inches in depth. Sand or anthracite is used singularly in mono-medium filter beds. Dual-medium beds may be comprised of anthracite and sand, activated carbon and sand, resin beads and sand, or resin beads and anthracite. In multi-medium beds some combination of anthracites, sand, garnet or ilmenite, activated carbon, and resin beads are used. In stratified filter beds, the effective size of the filter medium increases with the direction of wastewater flow. Flow through the filter medium can be either accomplished by gravity alone under pressure with the sometimes later described as rapid filters.

Several mechanisms are responsible for the removal of suspended solids in granular-medium filters. Included are straining, sedimentation, impaction, and interception. Chemical adsorption, physical adsorption, flocculation, and biological growth also may contribute to suspended solids removal. (Metcalf and Eddy, 1991).

The operation of granular-medium filters has two phases, filtration and cleaning or re-generation. The second phase, commonly called backwashing, involves the removal of captured suspended solids when effluent suspended solids begin to increase or when head loss across the filter bed reaches an acceptable maximum value. With semi-continuous filtration, filtration and backwashing occur sequentially, while with continuous filtration, the filtration and backwashing phases occur simultaneously. Usually backwashing is accomplished by reversing flow through the filter medium with sufficient velocity to expand or fluidize the medium to dislodge and transport accumulated suspended solids to the surface of the filter bed. Compressed air may be used in conjunction with the backwashing water to enhance removal of accumulated suspended solids. The backwashing water with the removed suspended solids typically is returned to a primary clarifier or a secondary biological treatment process unit (Metcalf and Eddy, 1991).

Filtration and backwashing occur simultaneously with continuous processes, and there is no suspended solids breakthrough or terminal head loss value. One type of continuous filter is the traveling bridge filter, which comprises a series of cells operated in parallel. Backwashing of individual cells occurs sequentially, while the other cells continue to filter influent. Deep bed filters, which are upflow filters, are continually backwashed by continually pumping sand from the bottom of the filter through a sand washing located at the top of the filter with the clean sand distributed on the top of the filter bed. Thus, sand flow is counter-current to the flow of the wastewater being filtered (Metcalf and Eddy, 1991). Generally, all types of granular-medium filter produce effluent with an average turbidity of two nephelometric turbidity units (NTUs) or less from high quality filter influent having turbidity of seven to nine NTUs. This level translates into a suspended solids concentration of 16 to 23 mg/L (Metcalf and Eddy, 1991). Lower quality filter influent requires chemical addition to achieve an effluent turbidity of two NTUs or less. Chemicals commonly used include a variety of organic polymers, alum, and ferric chloride. They produce removal of specific contaminants, including phosphorous, metal ions, and humic substances (Metcalf and Eddy, 1991).

Problems with the use of granular-medium filtration include turbidity breakthrough with semi-continuous filters, even though terminal head loss has not been reached. Problems with both semi-continuous and continuous filters include: buildup of emulsified grease; loss of filter medium, agglomeration of biological floc, dirt, and filter medium or media forming mud balls and reducing the effectiveness of filtration and backwashing, and the development of cracks in the filter bed (Metcalf and Eddy, 1991).

8.4.3 Alternate Tertiary Treatment Technologies

8.4.3.1 Nitrogen Removal

Besides the biological treatment discussed in Section 8.4.1.1, various physicochemical processes are used for nitrogen removal. The principal physical and chemical processes used for nitrogen removal are air stripping, breakpoint chlorination, and selective ion exchange. However, all these technologies are reported to have limited use due to cost, inconsistent performance, and operating and maintenance problems (Metcalf and Eddy, 1991; Johns, 1995). Air stripping and breakpoint chlorination is discussed in this section, while ion exchange is discussed in Section 8.4.3.3. Note that these three technologies remove nitrogen when the nitrogen is in the form of ammonia (air stripping, breakpoint chlorination, and ion exchange) or nitrate ions (ion exchange). Since, raw meat-processing wastewater contains nitrogen primarily in organic form, the technologies may require additional upstream treatment to convert the organic nitrogen into ammonia and/or nitrate.

Air Stripping

Air stripping of ammonia is a physical process of transferring ammonia from wastewater into air by injection of wastewater into air in a packed tower. To achieve a high degree of ammonia reduction, elevation of wastewater pH to at least 10.5 usually by the addition of lime, is necessary. The removal efficiencies of ammonia nitrogen can be as high as 98 percent with effluent ammonia concentrations of less than 1 mg/L (USEPA, 1974; USEPA, 1975). Because of the high operating and maintenance costs associated with air stripping, the practical application of air stripping of ammonia is limited to special cases, such as the need for a high pH for other reasons (Metcalf and Eddy, 1991).

High operation and maintenance costs for air stripping of ammonia can be attributed in part to the formation of calcium carbonate scale within stripping tower and feed lines. Absorption of carbon dioxide from the air stream used for stripping leads to calcium carbonate scale formation, which varies in nature from soft to very hard. Because the solubility of ammonia increases as temperature decreases, the amount of air required for stripping ammonia increases significantly as temperature decreases for the same degree of removal. If ice formation occurs in the stripping tower, a further reduction in removal efficiency occurs (Metcalf and Eddy, 1991, Johns, 1995).

There are secondary environmental impacts also because air stripping of ammonia without subsequent scrubbing in an acid solution results in the emission of ammonia to the atmosphere. This emission may lead to bad odor and air pollution. Particulate matter is also formed in the atmosphere, following the reaction of ammonia with sulfate. In addition, stripping towers can be sources of emissions of volatile organic compounds and noise (Peavy, 1986; Metcalf and Eddy, 1991).

Breakpoint Chlorination

Breakpoint chlorination involves the addition of chlorine to wastewater to oxidize ammonia to nitrogen gas and other stable compounds. Breakpoint chlorination has been successfully used as a second, stand-by ammonia removal process for ammonia concentrations up to 50 mg/L (Green et al., 1981). Before chlorine reacts with ammonia, it first reacts with oxidizable substances present, such as Fe^{+2}, Mn^{+2}, H_2S, and organic matter to produce chloride ions. After meeting the immediate demand of the oxidizable compounds, excess chlorine react with ammonia to form chloramines. With increased chlorine dosage, the chloramines formed will be converted to nitrogen trichloride, nitrous oxide, and nitrogen gas. The destruction of chloramines occurs until the breakpoint chlorination point is achieved. After this point, free residual chlorine becomes available (Metcalf and Eddy, 1991). Therefore, the required chlorine dosage to destroy ammonia is achieved when breakpoint chlorination is reached. The overall reaction between chlorine and ammonia can be described by the following equation:

$$2NH_3 + 3HOCl \longrightarrow N_2 + 3H_2O + 3HCl$$

Stoichiometrically, the breakpoint reaction requires a weight ratio of 7.6 CL_2 to 1 NH_4^+-N, but in actual practice ratios of from 8:1 to 10:1 are common (Green et al., 1981). Process efficiencies consistently range between 95 and 99 percent. The process is easily adapted to complete automation, which helps assure quality and operational control (Reynolds, 1982). The optimal pH for breakpoint chlorination is between 6 and 7. Because chlorine reacts with water, forming hydrochloric acid, a pH depression to below 6 may occur with poorly buffered wastewaters. This drop increases chlorine requirements and slows the rate of reaction.

One advantage of breakpoint chlorination for ammonia removal is its relative insensitivity to temperature. Also, capital costs are small relative to other ammonia removal processes, such as ammonia stripping and ion exchange (Green et al., 1981). However, many organic compounds react with chlorine to form toxic compounds, including trihalomethanes and other disinfection by-products, which can interfere with beneficial uses of receiving waters. Thus, dechlorination is necessary. Both sulfur dioxide and carbon adsorption are used with dechorination, with sulfur dioxide being more common due to lower cost. Another disadvantage of breakpoint chlorination for nitrogen removal may be an undesirable increase in total dissolved solids (Metcalf and Eddy, 1991).

8.4.3.2 Residual Suspended Solids Removal

Besides granular-medium filtration systems microscreens may be used to achieve supplemental removals of suspended solids. This practice also provides further reduction in BOD. Microscreens involve solid-liquid separation a process in which liquid passes through a filter fabric to remove as much fine material as possible.

Microscreens

Microscreens are a surface filtration device used to remove a portion of the residual suspended solids from secondary effluents and from stabilization pond effluents. Microscreens are low speed, continually backwashed, rotating drum filters operating under gravity conditions. Typical filtering fabrics have openings of 23 or 35 µm and cover the periphery of the drum. Wastewater enters the open end of the drum and flows outward through the rotating screening cloth. The collected solids are backwashed into a trough located at the highest point within the drum and returned to primary or secondary treatment processes (Metcalf and Eddy, 1991).

Typical suspended solids removal is about 55 percent with a range of 10 to 80 percent. Some problems with microscreens include incomplete solids removal and an inability to handle fluctuations in suspended solids concentrations. Reducing drum rotational speed and decreasing frequency of backwashing can increase removal efficiency, but screening capacity is thereby reduced. Typical hydraulic loading rates and drums speeds respectively are 75 to 150 gallon/ft^2-min and 15 ft/min at 3-in. head loss to 115 to 150 ft/min at a 6-in. head loss (Metcalf and Eddy, 1991).

8.4.3.3 Removal of Organic Compounds and Specific Ions

Various advanced wastewater treatment processes are used for removing organic compounds and target ions from wastewater. Carbon adsorption process has been widely used to remove organic compounds from different types of wastewater. To remove target ions from wastewater, ion exchange process have been used. To prevent filter plugging and to ensure proper operation, granular activated carbon columns and ion exchange columns are usually preceded by filtration units.

Carbon Adsorption

Both granular and powdered activated carbon can be used to further reduce concentrations of organic compounds, including refractory compounds after secondary biological treatment. With granulated activated carbon (GAC), the adsorption process occurs in steps. Initially, organic matter moves from the bulk liquid phase to the liquid-solid interface by advection and diffusion. Next, diffusion of the organic matter through the macropore system of the granulated activated carbon occurs at adsorption sites in micropores and submicropores. Although adsorption also occurs on the surface and in the macro- and mesopores of activated carbon granules, the surface area of the micro- and submicropores greatly exceeds the surface areas of the granule and the macro- and mesopores. With powdered activated carbon (PAC), adsorption occurs primarily on the surface of the carbon particles (Weber, 1972; Metcalf and Eddy, 1991).

When the rate of adsorption equals the rate of desorption, the adsorptive capacity of the carbon has been reached and regeneration is necessary. GAC is regenerated easily by oxidizing the adsorbed organic matter in a furnace. About 5 to 10 percent of GAC is destroyed in the regeneration process and must be replaced (Metcalf and Eddy, 1991). Also, the adsorptive capacity of regenerated GAC is slightly less than that of virgin GAC. A major problem with the use of PAC is that regeneration methodology is not well defined.

A fixed bed reactor often is used for wastewater treatment using GAC. Flow is downward through the carbon column, which is supported by an under-drain system. There may be provision for backwashing and surface washing to limit head-loss due to the accumulation of particulate matter. Upflow and expanded bed columns also are used (Metcalf and Eddy, 1991). With biological wastewater treatment, PAC usually is added either to the basin or to the secondary clarifier effluent. In the "PACT' process, the PAC is added directly to the aeration basin (Metcalf and Eddy, 1991).

Tertiary treatment using activated carbon can remove up to 98 percent of colloidal and dissolved organics measured as BOD_5 and COD in a wastewater stream. Effluent BOD_5 concentrations may be as low as 2 to 7 mg/L with effluent COD concentrations in the range of 10 to 20 mg/L (Metcalf and Eddy, 1991).

Use of activated carbon is common in water treatment to remove organic compounds from raw water supplies responsible for color, taste, and odor problems. In the treatment of MPP wastewaters, the use of carbon adsorption is generally limited to tertiary treatment prior to wastewater reuse as potable water.

Ion Exchange

Ion exchange is a unit process in which ions of a given species are displaced from an insoluble exchange material (resin) by ions of a different species in solution. This process is most commonly used to soften water by removing calcium and magnesium ions. It is also used in industrial wastewater treatment for the recovery of valuable constituents, including precious metals and radioactive materials. It may be operated in batch or continuous mode. In a batch process, the resin is stirred with the water to be treated in a the reactor until reaction is complete. The spent acid is removed by settling, and subsequently is regenerated and reused. In a continuous process, the exchange material is placed in a bed or a packed column, and the water to be treated is passed through it. When the resin capacity is exhausted, the column is backwashed to remove trapped solids and then regenerated (Metcalf and Eddy, 1991). To maintain continuous operation, typically, two or more columns are used, so that when one of the columns is off-line (backwashing or regenerating) other column(s) are on-line (operational).

Although ion exchange is known to occur with a number of natural materials, there is a broad spectrum of synthetic exchange resins available. Synthetic resins consist of networks of hydrocarbon radicals with attached soluble ionic functional groups. The hydrocarbon radicals are cross-linked in a three dimensional matrix, with the degree of cross-linking imparting the ability to exclude ions larger than a given size. The nature of the attached functional groups largely determines resin behavior. There are four major classes of ion exchange resins: strongly acidic and weakly acidic cation exchange resins, and strongly basic and weakly basic anion resins. Strongly acidic resins contain functional groups derived from strong acids such as sulfuric acid (H_2SO_4), whereas functional groups of weakly acidic resins are derived from weak acids such as carbonic acid (H_2CO_3). Similarly, strongly basic resins contain functional groups derived from quaternary ammonium compounds, whereas functional groups of weekly basic resins are derived from weak base amines. The exchangeable counter ion of an acidic cation resin may be the hydrogen ion or some other monovalent cation, such as sodium. For a basic anion resin, the exchangeable counter ion may be the hydroxide ion or some other monovalent anion. The regenerant will be the corresponding acid, base, or simple salt (Weber, 1972).

The use of ion exchange in the treatment of MPP wastewaters is less common. The ion exchange technology may be used to remove ammonium ions from wastewater, nitrate ions from the nitrified wastewater, phosphorous, and/or to remove total dissolved solids from wastewater. The functional group to be used depends on the target ions (NH_4^+, NO_3^- , or other ions) to be removed.

To minimize head loss through ion exchange columns and possible resin fouling, ion exchange usually follows granular medium filtration and possibly carbon adsorption. In addition, special provisions are necessary for regeneration waste. Another waste stream requiring disposal is exhausted resin. Regeneration efficiency decreases with time and replacement becomes necessary to maintain process performance.

8.5 DISINFECTION

Disinfection destroys remaining pathogenic microorganisms and is generally required for all MPP wastewaters being discharged to surface waters. Chlorine injection is the most commonly used method for wastewater disinfection; however, use of ultraviolet light for disinfection is not uncommon (USEPA, 2001). Ozone injection and combinations of UV and ozonation are also attractive alternatives for disinfection.

8.5.1 Chlorination

The chemical reactions that occur when chlorine is added to wastewater have been described above in the discussion of breakpoint chlorination for ammonia removal. For disinfection, the objective is to add chlorine at a rate that results in a free chlorine residual to ensure that pathogen kill occurs. As discussed above, a free chlorine residual occurs only after reactions with readily oxidizable ions, organic matter, and ammonia are complete. Thus, chlorine requirements for disinfection depend on wastewater characteristics at the time of disinfection. The degree of mixing and contact time in a chlorine contact chamber are critical factors in the process of disinfection using chlorine. The most commonly used chlorine compounds used for wastewater disinfection are chlorine gas, calcium hypochlorite, sodium hypochlorite, and chlorine dioxide (Metcalf and Eddy, 1991). Chlorine dioxide is an unstable and explosive gas that requires special precautions.

As also was noted above in the discussion of breakpoint chlorination for ammonia removal (Section 8.4.3.1), dechlorination often is necessary to reduce effluent toxicity with sulfur dioxide addition being the most commonly used approach. Sulfur dioxide reacts with both free chlorine and chloramines with chloride ions, resulting primarily in the end production of chloride ions (Metcalf and Eddy, 1991).

8.5.2 Ozonation

Since ozone is chemically unstable, it decomposes to oxygen very rapidly after generation, and thus must be generated on site. The most efficient method of producing ozone is by electrical discharge. Ozone is generated either from air or pure oxygen, when a high voltage is applied across the gap of narrowly spaced electrodes. It is an extremely reactive oxidant, and it is generally believed that bacterial kill through ozonation occurs directly because of cell wall disintegration. Ozone is a more effective virucide than chlorine. Ozone does not produce dissolved solids and is not affected by ammonia concentrations or pH. In addition, there is no chemical residue produce from using ozone, because it decomposes rapidly to oxygen and water. Use of ozone increases the dissolved oxygen concentration, control odor, and provides removal of soluble refractory organics. One disadvantage to using ozone is that it is necessary to generate it on site, because of its chemical instability (Metcalf and Eddy, 1991).

8.5.3 Ultraviolet Light

Suspended or submerged lamps producing ultraviolet (UV) light are another option for wastewater disinfection, especially for the inactivation of the parasites of *Cryptosporidium* parvum and *Giardia lamblia*. It is known that chlorine does not have an effect on Cryptosporidium and that ozone requires higher doses to complete inactivation (Stone and Brooks, 2001). Radiation emitted from the ultraviolet light is an effective bateriocide and virucide while generating any toxic compound. Low-pressure mercury arc lamps are the principal means of generating UV energy used for disinfection. Operationally the lamps are either suspended outside of the liquid to be treated or submerged in the liquid. Where the lamps are submerged, they are encased in quartz tubes to prevent cooling effects on the lamps. Radiation from low-pressure lamps with a wavelength of around 254 nm penetrates the cell wall of the microorganisms and is absorbed by cellular materials a process which either prevents replication or causes death of the cell to occur (Stone and Brooks, 2001). Since turbidity will absorb UV energy and shield the microorganism, turbidity in the water should be kept low for better results (Metcalf and Eddy, 1991). UV irradiation, whether at low- or medium-pressure, performs similarly in achieving 4 log inactivation of Cryptosporidium (Stone and Brooks, 2001). UV irradiation in combination with ozonation can also be applied for the reuse of chiller water in poultry operations (Diaz and Law, 1997).

8.6 EFFLUENT DISPOSAL

The most common disposal methods of treated MPP wastewaters are by discharge to adjacent surface waters under the authority of a NPDES permit or discharge to POTWs. However, disposal by land application is an alternative method that can eliminate the need for tertiary treatment of wastewater (Johns, 1995; Uhlman, 2001).

Land application by sprinkler or flood irrigation can be a feasible alternative to surface water discharge, if the appropriate land is available and other prerequisites can be satisfied. These prerequisites include soils with moderately slow to moderately rapid permeability and soils with the ability to collect any surface runoff that occurs. In addition, the production of a marketable crop is a necessity to provide a mechanism for the removal of nitrogen, phosphorus, and other nutrients from soils applied with wastewater by sprinkler or flood irrigation (Uhlman, 2001).

In land application, wastewater disposal is performed using a combination of percolation and evapotranspiration with microbial degradation of organic compounds occurring in the soil profile. Both crop uptake and nitrification-denitrification serve as mechanisms for nitrogen reduction. Crop uptake, chemical precipitation, and adsorption to soil particles are mechanisms of phosphorus reduction. Water balances are managed to match crop water use and salt leaching needs with irrigation to maintain water percolation to groundwater within the system design (Uhlman, 2001). Nitrogen balances are also developed to match estimated nitrogen losses and crop uptake (removal) to minimize percolate nitrate losses to groundwater. Spray and flood irrigation systems for wastewater disposal (Figure 8-4) may be designed with the objective of either wastewater disposal or wastewater reuse. If disposal is the objective, application or hydraulic loading rate is not controlled by crop requirements, but by the limiting design parameter, soil permeability or constituent loading. In many situations, nitrogen loading rate is the limiting design parameter to minimize leaching of nitrate nitrogen to ground water. Phosphorus loading rate generally is not a limiting design parameter, due to the ability of soils to immobilize phosphorus. However, the ability of soils to adsorb phosphorus is finite, and saturation of the upper zone of the soil profile can occur (US EPA, 1974).

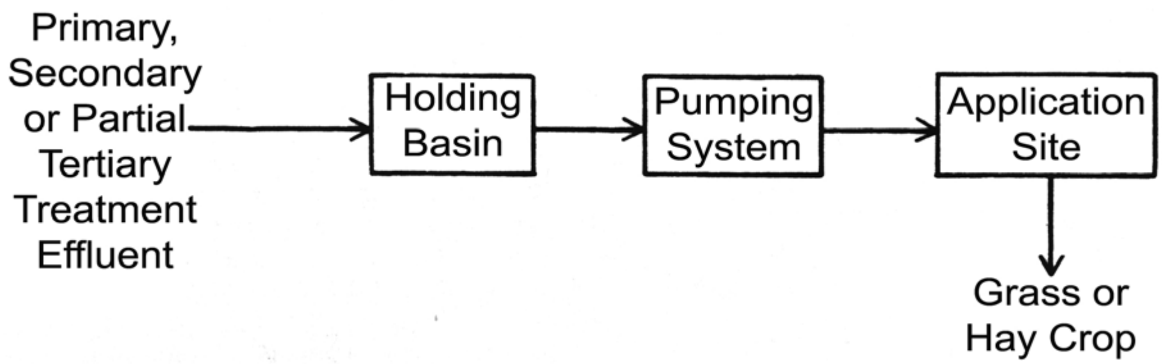

Figure 8-4. Spray/Flood Irrigation System (USEPA, 1974)

Wastewater can be applied to crops using solid set or center pivot sprinkler or flood irrigation. With flood irrigation, also known as ridge-and-furrow irrigation, wastewater is released into furrows between rows of growing crops. Fields irrigated using flood irrigation are graded to allow uniform irrigation of the entire field by gravity flow, with provision for capture and containment of any return flow. Intermittent application cycles, usually ranging from every four to ten days, maintain aerobic conditions in the soil. In arid and semi-arid areas, land application, as a method for wastewater disposal, is especially attractive, given the low rates of precipitation allowing higher hydraulic loading rates than in more humid regions. However, the accumulation of soluble salts (total dissolved solids) in the root zone of the soil profile can be problematic in arid and semi-arid regions because of the lack of precipitation, resulting in reduced leaching of these salts from the soil profile. These salt accumulations are toxic to many plant species. Salt accumulations in the soil profile also occur when conventional irrigation practices are used in arid and semi-arid climates. The typical approach used to deal with accumulations of soluble salts from irrigation is periodic hydraulic loadings to leach accumulated soluble salts from the root zone of the soil. However, some ground water contamination may result from using periodic hydraulic loadings. Reduction of total dissolved solids concentrations in MPP wastewaters prior to land application is another option, but the associated cost may make direct discharge to surface waters a more attractive option in arid and semi-arid climates.

Wastewater treatment systems using sprinkler or flood irrigation as a method for MPP wastewater disposal should provide a minimum of secondary treatment before use of wastewater for irrigation. Secondary treatment of wastewater reduces BOD and suspended solids loading rates and consequently, it reduces the potential of these parameters to act as limiting design factors. Secondary treatment also reduces the odor and vermin problems associated with flood irrigation or sprinkler application of lesser treated wastewater. A holding basin is a necessary element to allow intermittent wastewater applications and to provide storage when climatic or soil conditions do not allow irrigation. Ideally, storage should be adequate to limit wastewater application to the active plant growth period of the year. Thus, storage of wastewater for at least six months in cold climates is desirable (Loehr et al., 1979). For a more complete discussion of wastewater disposal by land application, Loehr et al. (1979) and Overcash and Pal (1979).

In the absence of proper system design and operation, land application as a method of wastewater disposal can adversely affect surface and ground water quality. Excessive organic loading rates can result in reduced soil permeability and the generation of noxious odors due to the development of anaerobic conditions. Excessive nitrogen application rates can lead to nitrate leaching to ground water. Excessive phosphorus application rates can lead to surface or ground water contamination, or both, if the irrigated soils become saturated with phosphorus.(Metcalf and Eddy, 1991)

Exposure to pathogens also is a concern, especially with spray irrigation systems given the potential for pathogen transport in aerosols. Virus transmission through aerosols is the most serious concern, because a single virus can cause infection. In contrast, infectious doses of bacterial pathogens range from at least 10^1 for *Shigella* to as high as 10^8 organisms for enteropathogenic *E. coli* (Loehr *et al*., 1979). However, using one or more of several recommended practices can reduce the transmission of pathogens in aerosols. Recommended practices include: (1) creating buffer zones with or without hedgerows (2) using low pressure nozzles aimed downward (3) avoiding wastewater spraying windy conditions and (4) restricting irrigation to daylight hours (Johns, 1995).

Especially in colder climates, wastewater land application systems require storage facilities to avoid application to frozen, snow-covered, or saturated soil. Wastewater application under these conditions can result in surface runoff transporting pollutants to adjacent surface waters. See Loehr et al (1979) for a detailed discussion of storage requirements for wastewater land application systems in various climates.

8.7 SOLIDS DISPOSAL

Typically, biosolids generated during the treatment of MPP wastewaters are aerobically digested before disposal by land application. Biosolids may be de-watered prior to land application. Rendering is a common disposal method for wastewater solids recovered by dissolved air flotation (DAF) before secondary treatment. Generally, the use of metal salts prior to DAF is avoided if rendering is used for the disposal of recovered solids, to unacceptably high concentrations of aluminum or iron in rendering products. Alternatives to rendering for the disposal of DAF solids are land application and land filling. High quality by-products (e.g., blood) are often segregated from DAF solids and other MPP WWTP sludges as some rendering operations (e.g., pet food manufacturing) require high quality input by-products.

EPA noted during site visits to two independent rendering operations that sludges from dissolved air floatation units which use chemical additions to promote solids separation are rendered; however, the chemical bond between the organic matter and the polymers requires that the sludges be processed (rendered) at higher temperatures (260 ^{o}F) and longer retention times. EPA also observed during site visits that some independent renderers reject raw materials that have (1) a pH below 4 SU (with 3 SU being a general cut-off), (2) ferric chloride due to its corrosive nature, and (3) other contamination (e.g., pesticides).

8.8 POLLUTION PREVENTION AND WASTEWATER REDUCTION PRACTICES

8.8.1 Wastewater Minimization and Waste Load Reduction Practices at MPP Facilities

For many MPP facilities, wastewater flow minimization and waste load reduction practices have been incorporated into normal business practices in order to reduce production costs and maximize profits. As with other competitive industries, unessential consumption of water and energy, and the additional cost~~s~~ of waste treatment can mean the difference between profitability and operational losses. While water reuse and by-product~~s~~ recovery are standard approaches for wastewater flow minimization and waste load reduction at MPP facilities, the extent of these practices and their effectiveness, varies widely among individual facilities. Some large facilities have installed onsite advanced wastewater treatment systems which treat facility effluent allowing this water to be reused for some applications within the facility. Other facilities have changed sanitation practices to reduce water use and effluence in general. For example, one independent renderer noted during an EPA site visit that his facility fully converted from a wet cleaning method to a dry cleaning method in the product shipment area in order to minimize water pollution.

Industry sources have estimated that the implementation of the U.S. Department of Agriculture Food Safety and Inspection Service's (USDA FSIS) Hazard Analysis and Critical Control Points (HACCP) program has increased water usage by 20 to 25 percent. USDA FSIS disagrees with industry's assertion that implementation of HACCP has necessarily required greater use of water. Furthermore, USDA FSIS asserts that its regulatory performance standards provide for numerous water reuse opportunities (see 9 CFR 416.2(g)).

The USDA FSIS promulgated the HACCP program on July 25, 1996 (61 FR 38806). The HACCP rule requires all MPP facilities to develop and implement a system of preventative controls to improve the safety of their products with an emphasis on reducing microbial contamination from fecal material. The Sanitation Requirements for Official Meat and Poultry

Establishments Rule (USDA, 1996; 64 FR 56400) also mandates all MPP facilities to develop and implement written standard operating procedures for sanitation.

As described below, opportunities remain for reducing potable water use and wastewater flow in MPP through water conservation techniques and multiple use and reuse of water. In addition, opportunities exist to reduce waste loads to wastewater treatment facilities by physically collected solid materials before using water to clean equipment and facilities. Gelman et al. (1989) and Berthouex et al. (1977) provide case studies for minimizing waste and water use at poultry processing and hog processing facilities, respectively. Both conclude that facilities can save costs through readily available process modifications that can significantly reduce water use and wastewater flow and loadings.

8.8.2 General Water Conservation and Waste Load Reduction Techniques

Reducing water use is important as facilities that institute a water use reduction program also reduce their raw wastewater load (Scaief, 1975). Numerous studies have demonstrated the water use in MPP can be reduced significantly. For example, Carawan and Clemens (1994) reported a reduction in water use of 75 gallons per pig processed, a reduction of 33 percent, following implementation of a water conservation program at a hog slaughtering and rendering operation. In addition, it has been demonstrated that substantial reductions in wastewater pollutant concentrations also can be achieved through implementation of waste load reduction practices. Reductions in 5-day biochemical oxygen demand (BOD_5) in hog processing wastewater of 40 percent have been reported (Carawan and Clemens, 1994). However, both goals can be achieved only when management recognizes that a reduction in processing costs and an increase in profitability can be realized by reducing the costs of potable water and wastewater treatment. Thus, a management commitment to water conservation logically depends on the cost of potable water, and a management commitment to waste load reduction depends on the cost of wastewater treatment. If potable water is being obtained from private on-site wells, there obviously is a reduced economic incentive to conserve water than when water is being purchased from a public utility or private water purveyor. Also, the incentive for waste load reduction generally is greater for indirect dischargers because wastewater treatment costs are readily identifiable and surcharges for excessive pollutant concentrations can rapidly escalate wastewater treatment costs. Conversely, wastewater treatment costs can be less visible for direct dischargers and less sensitive to pollutant concentrations.

The development of water conservation and waste load reduction programs in the MPP as well as in other industries begins with the development of general profiles of water use and wastewater pollutant concentrations over one or preferably several 24 hour periods to determine the relative significance of processing and cleanup activities. Generally this step is accompanied or followed by measuring water use in individual phases of the processing process to identify opportunities for water use reduction. For example, measuring water flow to scalders and chillers in poultry processing to determine overflow rates can identify overflow rates in excess of FSIS requirements. Measuring and regulating water pressure for carcass washing to insure that FSIS requirements are not being exceeded is another example of how water use can be reduced in MPP operations. Measuring and regulating small flows such as from hand washing operations also can significantly reduce water use and wastewater volume.

The daily cleanup and sanitation of processing facilities and equipment contributes substantially to water use and wastewater pollutant load and probably presents the greatest opportunity for reductions. Typically, both water use and wastewater pollutant load can be reduced substantially by initially “dry cleaning” processing areas and equipment to collect meat scraps and other materials for disposal by rendering instead of the common practice of using “water as a broom.” Although subsequent screening before wastewater treatment provides for recovery of larger particles, fine particulate matter and soluble proteins, fats, and carbohydrates are not recovered and are manifested as an increased pollutant load to the wastewater treatment plant. Gelman *et al.* (1989) have shown that biochemical oxygen demand (BOD) in cleanup wastewater in poultry processing can be reduced from 20 to 50 percent by initially dry cleaning processing areas and equipment. Concurrently, dry cleaning can increase the production of inedible rendered products. Dry cleaning of live animal holding areas also can reduce water required for the cleaning of these facilities and the pollutant load in the wastewater generated. However, responses to the MPP detailed survey indicate that dry cleaning is a much more common practice at meat as compared to poultry processing facilities (47 percent for meat processing respondents versus 17 percent for poultry processing respondents).

To be successful, water conservation and waste load reduction plans must be implemented and performance monitored. Implementation requires employee training that should be continual and possibly the installation of new equipment such as hose nozzles and foot valves at hand wash stations that automatically shut off when not in use. Conversion to high pressure, low volume systems for carcass washing and general sanitation also can reduce water consumption. However, continual monitoring of water use and waste loads also is a necessity to avoid slippage in performance.

8.8.3 Multiple Use and Reuse of Water

USDA FSIS guidelines do not preclude the multiple use and reuse of water in MPP as practices to reduce potable water consumption and the discharge of treated wastewater. While it is obvious that acceptable multiple use and reuse strategies must avoid contact with products intended for human consumption, a significant fraction of the water used in MPP does not involve such contact.

The multiple use of water most commonly occurs in poultry processing. Witherow et al. (1978) report that water conservation through multiple reuse in poultry processing will be rewarded by savings in processing cost and reduced requirements for wastewater treatment. Examples include the use of scalder overflow to flume feathers from mechanical de-feathering equipment and the use of chiller overflow to flume inedible viscera to screens for recovery prior to rendering. Combination UV irradiation and ozonation can be effective treatment for this re-used poultry chiller overflow (Diaz and Law, 1997). These are examples of countercurrent recycling where water reuse is countercurrent to product flow.

In contrast to multiple use, water reuse requires treatment as a prerequisite with the degree of treatment determining how water can be reused. For example, reuse of wastewater after tertiary treatment to remove suspended solids and double disinfection, such as chlorination followed by ultraviolet light, is permissible for purposes where no contact with such as evaporative condenser cooling and holding lot, parking lot, and wastewater treatment plant cleaning.

With further treatment to meet drinking water standards using unit processes such as coagulation and flocculation followed by settling and then filtration and disinfection, reuse of wastewater treatment plant secondary effluent expands the potential for reuse. Examples of permissible uses in hog processing include use on the kill floor up to the first carcass wash, flushing of large intestines (chitterlings), cleaning of receiving pens, and rendering facilities. Other possible uses of wastewater treated to meet drinking water standards include use for equipment such as pump cooling and as boiler makeup water.

In the poultry processing industry, a number of unit process level reuse strategies also have been explored. One example is the reuse of final chiller overflow following diatomaceous earth filtration and disinfection as scalder makeup water or for fluming of harvested giblets. As noted by Carawan (1994), it also was demonstrated in the late 1970s that poultry processing wastewater treated to meet primary drinking water standards can be safe, when mixed with an equal amount of potable water, for use in poultry processing.

Based on data provided by the MPP detailed survey, EPA estimates that reuse of water in MPP facilities is relatively rare. About 8 percent of the poultry processing respondents to the survey indicated reuse of water from the wastewater treatment plant to defeathering or evisceration areas. Other water reuse practices such as reusing effluent for screen washing or cleanup of outside areas are even less common as indicated by detailed survey response.

8.8.4 Specific Pollution Control Practices Identified by EPA in Previous Regulatory Proposals

The following relevant Best Available Technology Economically Achievable (BAT) in-plant pollution control practices were listed in EPA's "Development Document for Proposed Effluent Limitations Guidelines for the Poultry Segment of the Meat Product and Rendering Process Point Source Category" (USEPA, 1975):

- Control and minimize flow of freshwater at major outlets by installing properly sized spray nozzles and by regulating pressure on supply lines. Hand washers may require installation of press-to-operate valves. This also implies that screened waste waters are recycled for feather fluming.

- Confine bleeding and provide for sufficient bleed time. Recover all collectable blood and transport to rendering in tanks rather than by dumping on top of feathers or offal.

- Use minimum USDA-approved quantities of water in the scalder and chillers.

- Shut off all unnecessary flow during worm breaks.

- Consider the reuse of chiller water as makeup water for the scalder. This may require preheating the chiller water with the scalder overflow water by using a simple heat exchanger.

- Use pretreated poultry processing waste waters for condensing all cooking vapors in onsite rendering operations.

- Consider dry offal handling as an alternative to fluming. A number of plants have demonstrated the feasibility of dry offal handling in modern high-production poultry slaughtering operations.

- Consider steam scalding as an alternative to immersion scalding.

- Control water use in gizzard splitting and washing equipment.

- Provide for frequent and regular maintenance attention to byproduct screening and handling systems. A back-up screen may be required to prevent byproduct from entering municipal or private waste treatment systems.

- Dry clean all floors and tables prior to washdown to reduce the waste load. This is particularly important in the bleeding, cutting, and further processing areas and all other areas where there tend to be material spills.

- Use high-pressure, low-volume spray nozzles or steam-augmented systems for plant washdown.

- Minimize the amount of chemicals and detergents to prevent emulsification or solubilization of solids in the waste waters. For example, determine the minimum effective amount of chemical for use in the scald tank.

- Control inventories of raw materials used in further processing so that none of these materials are ever wasted to the sewer. Spent raw materials should be routed to rendering.

- Treat separately all overflow of cooking broth for grease and solids recovery.

- Reduce the waste water from thawing operations.

- Make all employees aware of good water management practices and encourage them to apply these practices.

- Treat offal truck drainage before sewering. One method is to steam sparge the collected drainage and then screen.

- In-plant primary systems—catch basins, skimming tanks, air flotation, etc.—should provide for at least a 30-minute detention time of the waste water. Frequent, regular maintenance attention should be provided.

The following BAT in-plant pollution control practices were listed in EPA's "Development Document for Proposed Effluent Limitations Guidelines and New Source Performance Standards for the Processor Segment of the Meat Products Point Source Category" (USEPA, 1974):

- Use water control systems and procedures to reduce water use considerable below that of Best Practicable Control Technology Currently Available (BPT) except for small processors.

- Reduce the waste water from thawing operations.

- Provide for improved collection and greater reuse of cure and pickle solutions.
- Prepackage products (e.g., hams) before cooking to reduce grease contamination of smokehouse floors and walls.
- Revise equipment cleaning procedures to collect and reuse wasted materials, or to dispose of them through channels other than the sewer.
- Reuse or recycle noncontaminated water whenever possible.
- Initiate and continually enforce meticulous dry cleanup of floors before washing.
- Install properly designed catch basins and maintain them with frequent regular grease and solids removal.

It should be noted that the in-plant controls and modifications required to achieve the July 1, 1983, effluent limitations included water control systems and procedures to reduce water use to about 50 percent of the water used to meet BPT (USEPA, 1974).

8.8.5 Non-Regulatory Approaches to Pollution Prevention

EPA is using non-regulatory approaches to facilitate reduction of wastewater generation in the MPP industry. Specifically, the Agency has formed partnerships with industry and state agencies to develop guidance materials and implement innovative practices for reducing waste.

Participants in developing this program include the American Meat Institute (AMI), the American Association of Meat Processors (AAMP), the U.S. Department of Agriculture (USDA), several State agencies, EPA programs and regions, and other interested constituent groups. For example, EPA and its partners are developing BMP guidance materials for handling and disposal of rendering materials, and for chloride, nitrogen, and phosphorus discharges. The project team will evaluate these management practices and develop measures of their effectiveness. Long-term deployment of the final tools will occur through the active leadership of the industry's trade associations. In addition, EPA is partnering with the Iowa Waste Reduction Center (IWRC) and the Iowa Department of Natural Resources (IDNR) to pilot test the Guide with five companies. IWRC and IDNR are providing technical assistance and implementation consulting to the five companies. The pilot will be completed in July 2002, and then EPA will evaluate the pilot and incorporate lessons learned into the final draft of the "EMS Guide for Meat and Poultry Processors." The final guide is expected to be completed by September 2002, at which point this tool will be widely marketed throughout the meat and poultry processing industry.

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