United States
Environmental Protection
Agency

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

The full document is available at: http://www.epa.gov/ost/guide/mpp/

EPA-821-R-04-011

SECTION 8

WASTEWATER TREATMENT TECHNOLOGIES AND POLLUTION PREVENTION PRACTICES

This section describes the 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 MPP wastewaters, EPA identified no practical difference in the types of treatment technologies between meat products 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 used in treating MPP wastewaters are similar to those normally used in treating domestic wastewaters (Eremektar et al., 1999; Johnston, 2001). In this section, the 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, phosphorus, or suspended solids or some combination thereof). MPP facilities that discharge directly to navigable waters under the authority of a National Pollutant Discharge Elimination System (NPDES) permit typically apply both primary and secondary treatment to generated wastewaters. As 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 Discharging Facilities Having the Treatment Unit in Place
Primary treatment	Screen	98
	Oil and grease removal	83
	Dissolved air flotation	81
	Flow equalization	75
Secondary and tertiary treatment	Biological treatment [a]	100
	Filtration	23
	Disinfection	92

Source: EPA detailed survey data.

[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).

8.1 PRIMARY TREATMENT

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; for example, flocculant or polymer is added to DAF units. Primary treatment has two objectives in the MPP industry: (1) to reduce suspended solids and biochemical oxygen demand (BOD) loads to subsequent unit processes, and (2) to recover materials that can be converted into marketable products through rendering.

8.1.1 Screening

Screening is typically the first and most inexpensive form of primary treatment. It 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). Several types of screens are used in wastewater treatment, including static or stationary, rotary drum, brushed, and 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 inch in diameter. Generally, all wastewater generated in MPP facilities is screened before discharge to subsequent treatment processes. The use of screens aids in recovering valuable by-products that are sometimes used as a raw material for the rendering industry and subsequent industries (Banks and Adebowale, 1991; USEPA, 1974, 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 it reduces the volume of solids to be recovered in subsequent unit processes, such as DAF (Starkey and Wright, 1997).

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

8.1.1.1 Static Screens

The primary function of a static screen (Figure 8-1) is to remove large solid particles (USEPA, 1974, 1975). For example, slaughterhouse raw wastewater can include coarse, suspended matter (larger than 1 mm mesh) that is insoluble, is slowly biodegradable, and accounts for 40 to 50 percent of the raw wastewater chemical oxygen demand (COD) (Johns, 1995). Screening can be accomplished in several ways. In older versions, only gravity drainage is involved. A concavely curved screen design that uses high-velocity pressure feeding and was originally developed for mineral classification has been adapted to meet MPP wastewater treatment needs. This design employs bar interference to the slurry, which slices off thin layers of the flow over the curved surface. The screen material is usually 316 stainless steel, although harder, wear-resistant stainless alloys can also be used for special purposes.

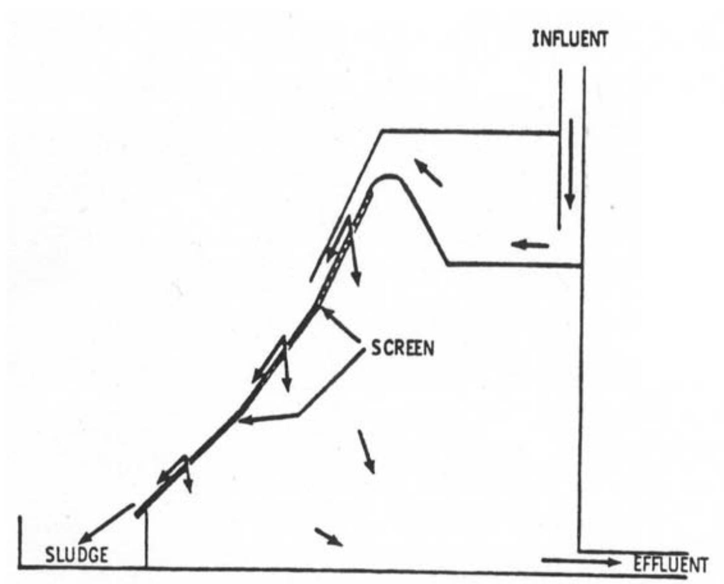

Figure 8-1. General schematic of a static screen (US EPA,1980).

Openings of 0.025 to 0.15 centimeter (0.01 to 0.06 inch) meet normal screening needs (USEPA, 1974, 1975).

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 feathers and solids from escaping 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 coarse particles before further screening by finer mesh screens.

8.1.1.2 Rotary Drum Screens

Rotary drum screens (Figure 8-2) are typically constructed of stainless steel mesh or wedge wire and are designed in one of two ways. In the first design the drum, 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, 1975).

The second type of rotary screen is driven by an external pinion gear. Raw wastewater discharges into the interior of the screen, below the center, and solids are removed in a trough mounted lengthwise with a screw conveyor. The liquid exits from the screen into a box, where the

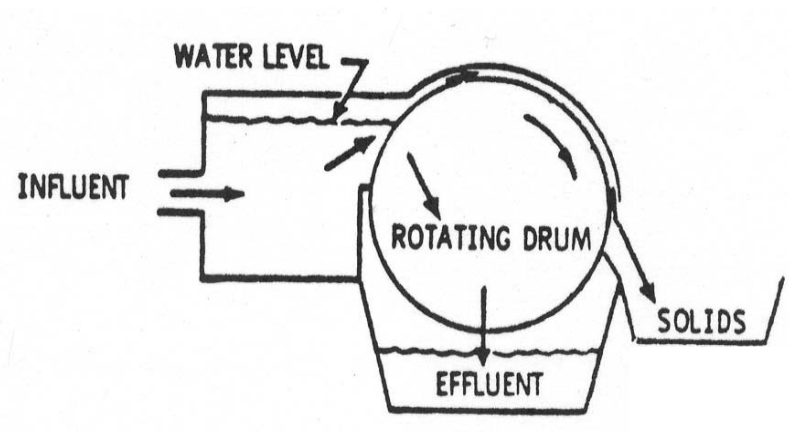

Figure 8-2. General schematic of a rotary drum screen (USEPA, 1980).

screen is partially submerged. The screen itself is typically 40 by 40 mesh, with openings of 0.4 millimeter. To assist in 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 of up to 82 percent have been reported (USEPA, 1974, 1975).

8.1.1.3 Brushed Screens

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).

8.1.1.4 Vibrating Screens

The effectiveness of a vibrating screen depends on rapid motion. Vibrating screens operate at between 99 and 1,800 revolutions per minute; the motion can be circular or straight, varying from 0.08 to 1.27 centimeters (1/32 to ½ inch) total travel. Speed and motion are selected by the screen manufacturer for the particular application (USEPA, 1974, 1975). Usually made of stainless steel, the vibrating screen 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 ensure long life. If the material is light or sticky however, the durability of the screening surface might be the least important factor. In such a case, a light wire might be desired to provide an increased percent of open area (USEPA, 1974, 1975).

Poultry products facilities use two types of vibrating screens. For offal recovery, vibrating 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. Vibrating screens with straight-line action are largely used for by-product recovery, while those with circular motion are frequently used for in-plant primary treatment (USEPA, 1975).

8.1.2 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 flow-through tank in which solids heavier than water sink to the bottom and grease and fine solids rise to the surface. A basin is equipped with a skimmer and a scraper. The skimmer moves grease and scum into collecting troughs, and the scraper moves sludge into a hopper. From the trough and hopper, the grease, scum, and sludge are pumped to by-product recovery systems. Key factors affecting basin efficiency are the detention time and the rate of solid removal from the basin. Depending on influent concentration, recovery rates of between 60 and 70 percent can be achieved with a detention time of 20 to 40 minutes (Nielsen, 1996).

Typically, catch basins are rectangular and relatively shallow. The preferred length is 1.8 meters or 6 feet. 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 1 peak hour with 30 to 40 minutes of detention. An equalization tank before the catch basin reduces size requirements significantly (USEPA, 1974, 1975). Depending on the influent characteristics, treatment costs range from $50 to $500 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 low overall maintenance and permanence of structure. Some facilities, however, 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, 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 those for steel; however, the all-steel tank requires a footing underneath and supporting members, whereas for the combined tank the concrete bottom forms the floor and supporting footings (USEPA, 1974, 1975).

8.1.3 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 time. Once particles reach the surface, they are removed by skimming (Metcalf and Eddy, 1991).

In DAF, 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 pounds per square inch (psi) (250 to 300 kilograms per __ (kPa), and then introduced into the flotation tank (Martin and Martin, 1991). The method of operation might cause operating costs to differ slightly, but process performance is essentially equal among the three modes of operation (USEPA, 1974, 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 such as polymers and flocculants are often added prior to the DAF system to improve its performance. Typical removals of suspended solids by DAF systems vary between 40 and 65 percent without chemical addition and between 80 and 93 percent with chemical addition. Likewise, oil and grease removal by a DAF system improves from 60 to 80 percent without chemical addition to 85 to 99 percent with chemical addition (Martin and Martin, 1991). A DAF system has many advantages, including its low installation cost, 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 parts as pumps and mechanical drives (USEPA, 1974, 1975).

Although alternatives to DAF 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 alternative technologies have prevented their incorporation (Johns, 1995). Cowan et al. (1992), however, 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 reported that ultrafiltration and reverse osmosis treatment might be the method of choice for treating slaughterhouse wastewaters, both as a pretreatment step prior to discharge to a publicly owned treatment works (POTW) and as a means of reclaiming high-quality reusable water from the treated effluent.

8.1.4 Flow Equalization

Because most MPP facilities operate on a 5-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 the 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 (Metcalf and Eddy, 1991; Reynolds, 1982). Flow equalization tanks can also be installed to store the effluent from the wastewater treatment plant before it is discharged to a

POTW or other effluent disposal destination. The end-of-treatment equalization ensures reduced variation in flow and waste load.

An equalization facility consists of a holding tank and pumping equipment designed to reduce the fluctuations of a waste stream. Such facilities can be economically advantageous, whether the industry is treating its own waste or discharging it 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 they may also be sized to provide a constant daily flow rate, even on non-processing days. The major advantages of equalization basins are that the subsequent treatment units are small, because they can be designed for the 24-hour average flow rather than peak flows, and that secondary waste treatment systems operate much better when not subjected to shock loads or variations in feed (USEPA, 1974, 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).

8.1.5 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 the DAF or clarifier to aggregate colloidal particles through destabilization by coagulation and flocculation to improve process performance. Essentially all 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 DAF units that use chemical addition 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 (127^{o}C or 260^{o}F) and for 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 and Valentine, 1992; USEPA, 1974, 1975).

8.2 SECONDARY BIOLOGICAL TREATMENT

MPP facilities that discharge directly to navigable waters under the authority of an NPDES permit at a minimum apply both primary and secondary treatment to generated wastewaters (see Table 8-1). The objective of secondary treatment is to reduce of BOD through the removal of the organic matter, primarily in the form of soluble organic compounds, that remains after primary treatment. Although secondary treatment of wastewater can be performed using a combination of physical and chemical unit processes, using biological processes has remained the preferred approach (Peavy, et al. 1986). Wastewater pollutant removal efficiencies of greater than 90 percent 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 that use 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.

8.2.1 Anaerobic Treatment

Anaerobic wastewater treatment processes use the microbially mediated reduction of complex organic compounds to methane and carbon dioxide as the mechanism for reducing organic matter and BOD. 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 it can be released directly to the atmosphere, collected and flared, or used as a boiler fuel (Clanton, 1997). USEPA (1997) provides estimates of the emission factors (e.g., gram-CH_4 per head of cattle) for these gases. The efficiency of BOD removal by anaerobic treatment can be very high. Anaerobic wastewater treatment processes are more sensitive than aerobic wastewater treatment processes to temperature and loading rate changes.

The production of biogas usually 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 by the reduction of compounds containing oxygen. A wide variety of facultative and anaerobic microorganisms are responsible for the transformations that occur to obtain energy for maintenance, growth, and nutrients, including carbon for cell synthesis (Metcalf and Eddy, 1991; Nielsen, 1996; Peavy et al., 1986).

In the second step, the alcohols and short-chained volatile acids are reduced further to methane and carbon dioxide by a group of obligate anaerobic microorganisms referred to collectively as methanogens. The methanogens include a number of species of methane-forming bacteria with growth rates significantly lower than those of 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 30 to 40 percent carbon dioxide and 60 to 70 percent methane plus trace amounts of hydrogen sulfide and other gases (Metcalf and Eddy, 1991; Nielsen, 1996; Peavy, 1986; Clanton, 1997).

Because of the negligible energy requirements of anaerobic wastewater treatment processes, these 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 the amount of energy required 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 treating MPP wastewaters. In addition to secondary treatment, anaerobic lagoons provide flow equalization. As noted previously, 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 (AF) processes (Johns, 1995). These alternatives are especially appealing in situations where land for lagoon construction or expansion is not available.

8.2.1.1 Anaerobic Lagoons

A typical anaerobic lagoon is relatively deep, 10 to 17 feet (3 to 5 meters), with a detention time of 5 to 10 days. Many treatment systems comprise at least two lagoons in parallel or series and typical loading rates are between 15 and 20 pounds BOD_5 1,000 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, 1975). Depending on the operating conditions, the BOD reductions by anaerobic lagoons can vary widely. Reductions up to 97 percent of BOD_5, up to 95 percent of suspended solids, and up to 96 percent of COD from the influent have been reported (John, 1995; USEPA, 1974, 1975).

Wastewater organic carbon anaerobic degradation products emitted from anaerobic lagoons include methane and carbon dioxide. Ammonium and hydrogen sulfide are also 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 the emissions produced in the anaerobic lagoons. A pH of 8 or greater causes more ammonia to be emitted; 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 (thereby 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 loss, which increases microbial reaction rates. Surface area loading rates can thus be increased and lagoon volume reduced (Morris et al., 1998).

8.2.1.2 Alternative Anaerobic Treatment Technologies

Anaerobic Contact System

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 maintain 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 kilograms per cubic meter 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 can be used to maintain the 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, and treatment occurs when the wastewater comes in contact with the granules. The methane and carbon dioxide produced generate internal circulation and maintain the floating sludge blanket. Biogas is collected in a gas collection dome 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 has been used to reduce unwanted filamentous growth (Eckenfelder, 1989).

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

Anaerobic Filter (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, the AF provides the ability to treat wastewaters with COD concentrations as high as 20,000 milligrams per liter (mg/L), as well as resistance to shock loads. Several studies have shown that AFs operated at short HRTs 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 kilograms per cubic meter per day (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 (91°F), 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 because 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 to MPP wastewaters because high protein concentrations eliminate the need for supplemental alkalinity. In addition, an ASBR easily addresses the high levels of solids 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 might necessitate at least partial removal of the hydrogen sulfide prior to use as a fuel (Morris et al., 1998).

8.2.2 Aerobic Treatment

In the treatment of MPP wastewaters, aerobic treatment might directly follow primary treatment. More typically, it follows some form of anaerobic treatment to reduce BOD and suspended solids concentrations to the levels required for discharge. Reduction of ammonia is also 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. The operating costs of aerobic systems, however, are higher than the costs of anaerobic systems, however, for processing livestock wastewater because of the relatively high space, maintenance, management, and energy requirements of artificial oxygenation. The microorganisms involved in the 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 the activated-sludge process, such as 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 use a diverse population of heterotrophic microorganisms that use molecular oxygen in the process of obtaining energy for cell maintenance and growth (Metcalf and Eddy, 1991).

The primary objective of aerobic wastewater treatment processes is transforming soluble and colloidal organic compounds into microbial biomass, with subsequent removal of the biomass by settling or mechanical separation as the primary mechanism for removal of organic matter and BOD. Some oxidation of organic carbon to carbon dioxide also occurs, providing energy for cell maintenance and growth. The degree of carbon oxidation depends on the 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 of BOD removed. Attached growth processes usually 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 depends on the type of treatment system used and its operating conditions.

8.2.2.1 Activated Sludge

The activated sludge process (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, the most common forms of the activated sludge process used in the MPP industry include conventional, complete mix, extended aeration, oxidation ditch, and sequencing batch reactor. Other forms of the process that are sometimes used 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 composed primarily of bacteria and protozoa, which aggregate to form flocs.

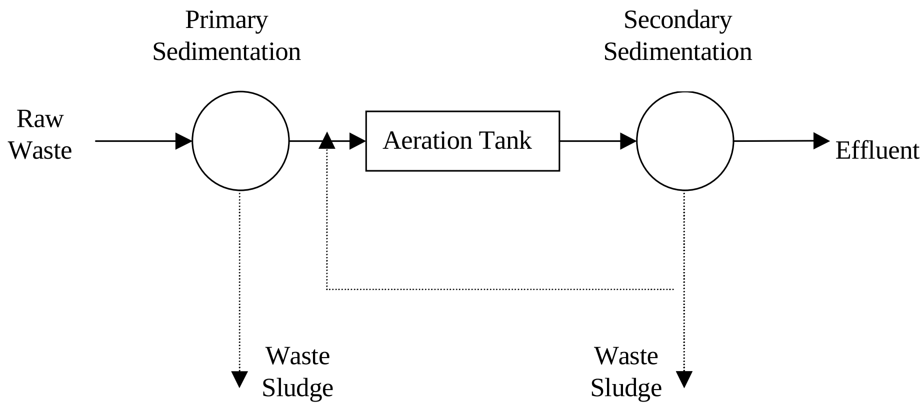

Figure 8-3. Activated Sludge Process (USEPA, 1974).

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 as 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 by using aerobic or anaerobic digestion or by adding chemicals (lime stabilization), which can be followed by dewatering by filtration or centrifugation (USEPA, 1974, 1975).

The activated sludge process is capable of 95 percent reductions in BOD_5 and suspended solids (USEPA, 1974, 1975). In addition, reductions in ammonia nitrogen in excess of 95 percent are possible at temperatures above 10 °C (50 °F) and dissolved oxygen concentrations above 2 mg/L (Johns, 1995). Performance depend on maintaining an adequate SRT and mixed liquor suspended solids with good settling characteristics, which depend on floc formation. Excessive growth of filamentous organisms can impair the settleability of activated sludge. 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. A plug flow regime can 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 the tank's length. During the aeration period, adsorption, flocculation, and oxidation of organic matter occur. Activated-sludge solids are separated in a secondary settling tank (Metcalf and Eddy, 1991).

Complete Mix

The 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 points in the aeration tank. The organic load on the aeration tank and the oxygen demand are uniform throughout the tank's 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 the 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, the poor settling characteristics of the aeration basin effluent are a frequently encountered problem with extended aeration. In general, 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 a high degree of nitrification.

Oxidation Ditches

The 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 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 feet per second (ft/s). Oxidation ditches typically operate in an extended aeration mode with an HRT greater than 10 hours and an SRT of 10 to 50 days (USEPA, 1993). Oxidation ditches provide high removal of BOD and can be designed for nitrification and nitrogen and phosphorus removal (Sen et al., 1990).

Sequencing Batch Reactor

The sequencing batch reactor (SBR) is a fill-and-draw reactor system that uses one or more complete mix tanks in which all steps of the 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). These periods may be modified or eliminated, however, depending on effluent requirements. The time for a complete cycle is the total time between the beginning of fill and the end of idle (Martin and Martin, 1991). SBR systems provide high removal of BOD and suspended solids. In addition, these systems can be designed for nitrification and removal of nitrogen and phosphorus. Lo and Liao (1990) reported that SBR technology can be used successfully in the treatment of poultry processing wastewaters for the removal of 5-day BOD (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).

8.2.2.2 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 8.2.1.1. 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 5 feet deep. High-rate ponds are usually shallower, with a maximum depth of 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 nonphotosynthetic 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 can be operated in parallel or in a series. To maximize performance, intermittent mixing is necessary. Without supplemental aeration, dissolved oxygen concentrations vary from supersaturation due to photosynthesis during daylight hours to values at or approaching zero at night, especially with high-rate ponds. In addition, without aeration, settled solids form an anaerobic zone at the bottom of the pond (Reynolds, 1982).

The low cost of aerobic stabilization ponds is offset, especially in colder climates, by seasonal variation in performance. In winter, limited sunlight due to cloud cover and shorter day length 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 can become anaerobic lagoons in winter months with a concurrent deterioration in effluent quality. They can also become 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 are typically about 8 feet (2.4 meters) deep but can be as much as 15 feet (4.6 meters) deep. They can 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 for 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 to recover settled solids for recycle. Aerated lagoons operated as complete mix systems without solids recycle might use a large, shallow, earthen basin in place of a more conventional clarifier for removing suspended solids. Typically, these basins are also 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 their relatively low capital cost; however, more land is required. With earthen settling basins, algae growth and odors, along with inconsistent effluent quality, can be problems.

Facultative Lagoons

Facultative lagoons are deeper than aerobic lagoons, varying in depth from 5 to 8 feet. 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 when dissolved oxygen is present and anaerobically otherwise. The facultative lagoons have more depth and smaller surface areas than aerated or aerobic lagoons. They still have good odor control capabilities, however, 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 facultative lagoons are a combination of aerobic and anaerobic degradation reactions (Zhang, 2001).

8.2.2.3 Alternate Aerobic Treatment Technologies

Trickling Filters

A trickling filter consists of a bed of highly permeable media to which microbial flora become attached, a distribution system to spread wastewater uniformly over the bed surface, and an under-drain system for collecting 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 into the film or slime layer of attached microorganisms. Within 0.1 to 0.2 millimeter 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. Anaerobic conditions develop near the media surface. In addition, organic matter and nutrients necessary for cell maintenance and growth are lacking because of 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 it can be a periodic or continual process depending on the organic and hydraulic loading rates. The hydraulic loading rate is usually 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 can include the nitrifying bacteria *Nitrosomonas* and *Nitrobacter*. It can also 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 square foot per-day for low-rate filters to 0.8 to 3.2 gallons per square foot per day for roughing filters. Organic loading rates range from 5 to 25 pounds BOD_5 per 10^3 square foot per day to 100 to 500 pounds BOD_5 per 10^3 square foot per day. Low-rate and two-stage trickling filters can produce a nitrified effluent, while roughing filters provide no nitrification. Others might 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 employ only rock. Super high-rate filters use plastic media, while roughing filters may be constructed using 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 is also necessary. Lower energy requirements make trickling filters attractive alternatives to activated sludge processes. Mass-transfer limitations, however, 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 can be as great as 95 percent (USEPA, 1974).

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 to the atmosphere for adsorption of oxygen. The rate of rotation controls oxygen diffusion into the attached microbial film and provides the shear 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 multistage 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 can be derived from pilot plant or full-scale performance evaluations or by 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 gal/ft^2/day and 2.0 to 3.5 pounds total $BOD_5/10^3$ square foot per 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 gal/ft^2/day and 1.5 to 3.0 pounds $BOD_5/10^3$ square foot per 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 multistage 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 sensitivity (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. The energy requirements associated with activated-sludge processes might 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).

8.3 TERTIARY TREATMENT

Tertiary or advanced wastewater treatment is usually 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 can be retrofit without the addition of new tanks or lagoons to incorporate biological nutrient removal (Randall et al., 1999). In addition, tertiary wastewater treatment can be used to remove soluble refractory, toxic, and dissolved inorganic substances. In the treatment of MPP wastewaters, tertiary wastewater treatment is most commonly used for further reductions in nutrients and suspended solids.

8.3.1 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. The

limited assimilative capacity of receiving waters, however, can require additional reductions in nitrogen and phosphorus concentrations before discharge. Both biological and physicochemical unit processes can be used to reduce nitrogen and phosphorus concentrations in wastewater. Biological processes are typically more cost effective than physicochemical processes. Moreover, retrofitting existing secondary treatment systems for biological nutrient removal can lead to reduced costs given the lower requirements for energy use and chemical addition (Randall and Mitta, 1998; Randall et al., 1999).

8.3.1.1 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, is also 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 can also 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 is usually 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 is probably the most commonly added source of organic carbon for denitrification, although raw wastewater (bypassed to the denitrification treatment tank), biosolids, and a variety of other substances also can be used (Metcalf and Eddy, 1991; USEPA, 1993).

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 to carbon dioxide and nitrate, respectively (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 can 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 is 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 that uses 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), A^2/O, and University of Capetown (UCT) processes (USEPA, 1993). The A^2/O and UCT processes were developed to remove both nitrogen and phosphorus. 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 retrofitting 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 total nitrogen in the discharged effluent year-round and (2) significantly reducing operational costs by reducing the need for electrical energy, aeration, and chemical addition (Sen et al., 1990).

Nitrification is easily inhibited by a number of factors, such as toxic organic and inorganic compounds, pH, and temperature. In poorly buffered systems, the hydrogen ions released when ammonia is oxidized to nitrite or 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^-) used is 8.64 milligrams HCO_3^- per milligram of ammonia nitrogen oxidized to nitrate nitrogen. For *Nitrosomonas*, the equation is

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

and 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 usually 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. Some organic matter, however, must be converted into cellular material and is not completely oxidized. Nevertheless, it does 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 than 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; the magnitude decreases as SRT increases. Assuming an

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 that occurs under anaerobic conditions. Typical cell yield under anaerobic conditions is 0.05 mg volatile suspended solids (VSS) per milligram BOD removed versus 0.6 milligram 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 might become harder and require excessively long SRTs in cold weather (Metcalf and Eddy, 1991; USEPA, 1993).

8.3.1.2 Phosphorus Removal

To achieve low effluent discharge limits, phosphorus can be removed from wastewater by using biological treatment and/or 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. Luxury uptake and storage for subsequent use of phosphorus occurs, however, 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 (Metcalf and Eddy, 1991; Reddy, 1998; USEPA, 1987).

Several proprietary processes use luxury uptake to remove phosphorus from wastewater during suspended-growth secondary treatment. Included are the A/O, PhoStrip, and Bardenpho processes. In addition, 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 is then 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 SBR is capable of combined biological removal of nitrogen and phosphorus (Metcalf and Eddy, 1991; Reddy, 1998; USEPA, 1987).

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. Ferrous sulfate and ferrous chloride can also be used. Use of lime is less common because of the operating and maintenance problems associated with its use and the large volume of sludge produced. Polymers are often used in conjunction with metal salts to improve the degree of phosphorus removal. Ion exchange, discussed in Section 8.4.3.3, is also an option for phosphate phosphorus removal, but it is rarely used in wastewater treatment (Metcalf and Eddy, 1991).

Chemicals can be added to remove phosphorus (1) in raw wastewater prior to primary settling, (2) in primary clarifier effluent, (3) in mixed liquor with suspended-growth treatment processes, (4) in 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 done 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 DAF 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, 1 mole theoretically will precipitate 1 mole of phosphate. However, competing reactions and the effects of alkalinity, pH, trace elements, and ligands found in wastewater make bench-scale or full-scale tests necessary to determine dosage rates. Because of coagulation and flocculation, suspended solids are also removed 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 might be necessary, depending on the wastewater's buffer capacity (Metcalf and Eddy, 1991; Reddy, 1998; USEPA, 1987).

When lime is used, it is usually calcium hydroxide ($Ca(OH)_2$). Because a reaction with natural bicarbonate alkalinity forms $CaCO_3$ 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 (Metcalf and Eddy, 1991; Reddy, 1998; USEPA, 1987).

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

8.3.2 Residual Suspended Solids Removal

Simple clarification after secondary wastewater treatment might 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 semi-continuous or continuous, depending on whether backwashing is a batch or a semi-continuous operation or a 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 filter medium. Shallow, conventional, and deep bed filters are typically about 11 to 16, 30 to 36, and 72 inches, respectively, in depth. Sand or anthracite is used alone in mono-medium filter beds. Dual-medium beds can be composed 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 is 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 accomplished by gravity alone or under pressure with the use of 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 can also 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 removing 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; with continuous filtration, the filtration and backwashing phases occur simultaneously Backwashing is usually accomplished by reversing flow through the filter medium with sufficient velocity to expand or fluidize the medium to dislodge accumulated suspended solids and transport them to the surface of the filter bed. Compressed air can 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 backwashed by continually pumping sand from the bottom of the filter through a sand wash at the top of the filter. The clean sand is distributed on the top of the filter bed. Thus, sand flow is countercurrent to the flow of the wastewater being filtered (Metcalf and Eddy, 1991). In general, all types of granular-medium filters produce effluent with an average turbidity of 2 nephelometric turbidity units (NTU) or less from high-quality filter influent having a turbidity of 7 to 9 NTU. This level translates to 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 2 NTU or less. Chemicals commonly used include a variety of organic polymers, alum, and ferric chloride. They remove specific contaminants, including phosphorus, metal ions, and humic substances (Metcalf and Eddy, 1991).

Problems with the use of granular-medium filtration include turbidity breakthrough with semi-continuous filter 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 the media’s formation of mud balls and reduction of the effectiveness of filtration and backwashing; and development of cracks in the filter bed (Metcalf and Eddy, 1991).

8.3.3 Alternative Tertiary Treatment Technologies

8.3.3.1 Nitrogen Removal

In addition to the biological treatment discussed in Section 8.3.1.1, various physicochemical processes are used to remove nitrogen. The principal physical and chemical processes used for nitrogen removal are air stripping, breakpoint chlorination, and selective ion exchange. All these technologies, however, are reported to have limited use because of their cost, inconsistent performance, and operating and maintenance problems (Johns, 1995; Metcalf and Eddy, 1991). Air stripping and breakpoint chlorination are discussed in this section, and ion exchange is discussed in Section 8.3.3.3. 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). Because raw meat-processing wastewater contains nitrogen primarily in organic form, the technologies might require additional upstream treatment to convert the organic nitrogen into ammonia or nitrate.

Air Stripping

Air stripping of ammonia is a physical process of transferring ammonia from wastewater into air by injecting the wastewater into air in a packed tower. To achieve a high degree of ammonia reduction, elevating the wastewater pH to at least 10.5, usually by adding 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, 1975). Because of the high operation and maintenance costs associated with air stripping, the practical application of air stripping of ammonia is limited to special cases, such as those where a high pH is needed for other reasons (Metcalf and Eddy, 1991).

The high operation and maintenance costs for air stripping of ammonia can be attributed in part to the formation of calcium carbonate scale within the stripping tower and feed lines. Absorption of carbon dioxide from the air stream used for stripping leads to calcium carbonate scale formation. The scale 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 removal efficiency is further reduced (Johns, 1995; Metcalf and Eddy, 1991).

Secondary environmental impacts also occur because air stripping of ammonia without subsequent scrubbing in an acid solution results in the emission of ammonia to the atmosphere. This emission can lead to unpleasant odors and air pollution. Particulate matter is also formed in the atmosphere, following the reaction of ammonia with sulfate. In addition, stripping towers can emit volatile organic compounds and cause noise (Peavy et al., 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. This technology 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 the 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 reacts with ammonia to form chloramines. With increased chlorine dosage, the chloramines formed are 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 ensure 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 might occur with poorly buffered wastewaters. Such a drop increases chlorine requirements and slows the rate of reaction.

One advantage of breakpoint chlorination for ammonia removal is its relative insensitivity to temperature. In addition 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. Therefore, dechlorination is necessary. Both sulfur dioxide and carbon adsorption are used for dechorination; sulfur dioxide is the more common because of its lower cost. Another disadvantage of breakpoint chlorination for nitrogen removal is the potential for an undesirable increase in total dissolved solids (Metcalf and Eddy, 1991).

8.3.3.2 Residual Suspended Solids Removal

Microscreens can also be used to achieve supplemental removal 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 surface filtration devices 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 that operate under gravity conditions. Typical filter fabrics have openings of 23 or 35 micrometers 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 in the drum and returned to primary or secondary treatment processes (Metcalf and Eddy, 1991).

Typical suspended solids removal is about 55 percent; the range is 10 to 80 percent. Some problems with microscreens are 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 drum speeds are 75 to 150 gal/ft^2/min and 15 ft/min at a 3-inch head loss to 115 to 150 ft/min at a 6-inch head loss (Metcalf and Eddy, 1991).

8.3.3.3 Removal of Organic Compounds and Specific Ions

Various advanced wastewater treatment processes are used for removing organic compounds and target ions from wastewater. The carbon adsorption process has been widely used to remove organic compounds from different types of wastewater. To remove target ions from wastewater, ion exchange processes 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 areas of the micro- and submicropores greatly exceed 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 (Metcalf and Eddy, 1991; Weber, 1972).

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 the regeneration methodology is not well defined.

A fixed-bed reactor is often used for wastewater treatment using GAC. Flow is downward through the carbon column, which is supported by an under-drain system. There might be provision for backwashing and surface washing to limit head loss due to the accumulation of particulate matter. Upflow and expanded bed columns are also used (Metcalf and Eddy, 1991). With biological wastewater treatment, PAC is usually added 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 can 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 to recover 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 the reactor until reaction is complete. The spent acid is removed by settling and is subsequently 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), the other column(s) are on-line (operational).

Although ion exchange is known to occur with a number of natural materials, a broad spectrum of synthetic exchange resins are 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, or phosphorus, or 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.

8.4 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 (UV) light for disinfection is not uncommon (USEPA, 2001). Ozone injection and combinations of UV and ozonation are also attractive disinfection alternatives.

8.4.1 Chlorination

The chemical reactions that occur when chlorine is added to wastewater have been described 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 previously, a free chlorine residual occurs only after reactions with readily oxidizable ions, organic matter, and ammonia are complete. Therefore, 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 chlorine compounds most commonly 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 handling and safety precautions.

As also noted in the discussion of breakpoint chlorination for ammonia removal (Section 8.4.3.1), dechlorination is often necessary to reduce effluent toxicity. Sulfur dioxide addition is 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).

8.4.2 Ozonation

Because 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 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, no chemical residue is produced by using ozone because ozone decomposes rapidly to oxygen and water. Using ozone increases the dissolved oxygen concentration, controls odor, and provides removal of soluble refractory organics. One disadvantage of using ozone is that it must be generated on-site because of its chemical instability (Metcalf and Eddy, 1991).

8.4.3 Ultraviolet Light

Suspended or submerged lamps producing UV light are another option for wastewater disinfection, especially for the inactivation of the parasites *Cryptosporidium parvum* and *Giardia lamblia*. It is known that chlorine does not have an effect on *Cryptosporidium* and that high doses of ozone are required to complete inactivation (Brooks and Stone, 2001). Radiation emitted from the UV light is an effective bactericide and virucide that does not generate any toxic compound. Low-pressure mercury arc lamps are the principal means of generating the UV energy used for disinfection. Operationally, the lamps are either suspended outside 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 nanometers penetrates the cell wall of the microorganisms and is absorbed by cellular materials in a process that prevents replication or causes death of the cell (Stone and Brooks, 2001). Turbidity in the water absorbs UV energy and shields the microorganisms, and therefore it should be kept low for better results (Metcalf and Eddy, 1991). UV irradiation, whether at low or medium pressure, performs similarly in achieving a 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).

8.5 EFFLUENT DISPOSAL

The most common disposal methods for treated MPP wastewaters are discharge to adjacent surface waters under the authority of an NPDES permit or discharge to POTWs. Disposal by land application, however, 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 necessary to provide a mechanism for the removal of nitrogen, phosphorus, and other nutrients from the soils to which wastewater has been applied (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 (removal) and nitrification-denitrification are mechanisms of 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 ground water within the system design (Uhlman, 2001). Nitrogen balances are also developed to match estimated nitrogen losses and crop uptake to minimize percolate nitrate losses to ground water. Spray and flood irrigation systems for wastewater disposal (Figure 8-4) can be designed with the objective of either wastewater disposal or wastewater reuse. If disposal is the objective, the application or hydraulic loading rate is controlled not 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 is not usually a limiting design parameter because of the ability of soils to immobilize

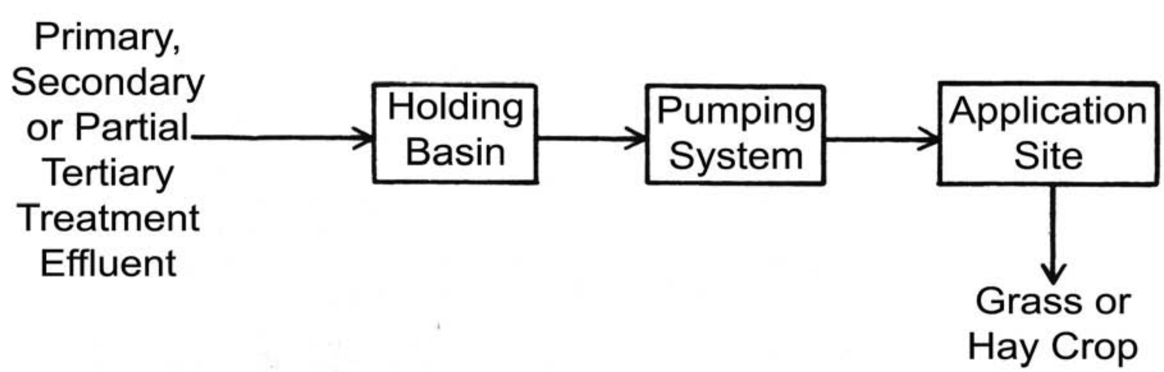

Figure 8-4. Spray/Flood Irrigation System (USEPA, 1974).

phosphorus. The ability of soils to adsorb phosphorus is finite, however, and saturation of the upper zone of the soil profile can occur (USEPA, 1974).

Wastewater can be applied to crops using solid set or center pivot sprinklers 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 every 4 to 10 days, maintain aerobic conditions in the soil. In arid and semiarid areas, land application as a method for wastewater disposal is especially attractive because the low rates of precipitation allow 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. Such 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 semiarid 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, although some ground water contamination might result. Reduction of total dissolved solids concentrations in MPP wastewaters prior to land application is another option, but the associated cost might make direct discharge to surface waters a more attractive option in arid and semiarid climates.

Wastewater treatment systems using sprinkler or flood irrigation as a method for MPP wastewater disposal should provide at least secondary treatment before using the wastewater for irrigation. Secondary treatment of wastewater reduces BOD and suspended solids loading rates and thereby 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 less-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 6 months in cold climates is desirable (Loehr et al., 1979). For a more complete discussion of wastewater disposal by land application, refer to 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 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 is also 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 organisms for *Shigella* to as high as 10^8 organisms for enteropathogenic *E. coli* (Loehr et al., 1979). Using one or more of several recommended practices, however, can reduce the transmission of pathogens in aerosols. Those practices include (1) creating buffer zones with or without hedgerows, (2) using low-pressure nozzles aimed downward, (3) avoiding wastewater spraying under 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. Refer to Loehr et al. (1979) for a detailed discussion of storage requirements for wastewater land application systems in various climates.

8.6 SOLIDS DISPOSAL

Typically, biosolids generated during the treatment of MPP wastewaters are aerobically digested before disposal by land application. Biosolids may be dewatered before land application. Rendering is a common disposal method for wastewater solids recovered by 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 because of the potential for 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 wastewater treatment plant (WWTP) sludges because some rendering operations (e.g., pet food manufacturing) require high-quality by-products as input.

EPA noted during site visits to two independent rendering operations that sludges from DAF units that 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 (with 3 being a general cutoff), (2) ferric chloride due to its corrosive nature, and (3) other contamination (e.g., pesticides).

8.7 POLLUTION PREVENTION AND WASTEWATER REDUCTION PRACTICES

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

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 preventive 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 that all MPP facilities develop and implement written standard operating procedures for sanitation.

As described below, opportunities remain for reducing potable water use and wastewater flow in MPP facilities 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 collecting solid materials before using water to clean equipment and facilities. Gelman et al. (1989) and Berthouex et al. (1977) provide case studies of 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, wastewater flow and loadings.

8.7.2 General Water Conservation and Waste Load Reduction Techniques

Reducing water use is important because facilities that institute a water use reduction program also reduce their raw wastewater load (Scaief, 1975). Numerous studies have demonstrated that water use in MPP facilities can be reduced significantly. For example, Carawan and Clemens (1994) reported a reduction in water use of 75 gallons per pig processed, a 33 percent reduction, after a water conservation program was implemented at a hog slaughtering and rendering operation. In addition, it has been demonstrated that substantial reductions in wastewater pollutant concentrations can be achieved by implementing waste load reduction practices. Reductions in 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. When potable water is being obtained from private on-site wells, there is obviously less economic incentive to conserve water than when water is being purchased from a public utility or private water purveyor. In addition, 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 industry, 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. This step is usually accompanied or followed by measuring water use in individual phases of the processing process to identify opportunities for reducing water use. For example, measuring water flow to scalders and chillers in poultry processing to determine overflow rates can identify rates in excess of the FSIS requirements. Measuring and regulating water pressure for carcass washing to ensure that the 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 those from hand-washing operations can also 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 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 can also reduce the amount of water required for the cleaning these facilities and the pollutant load in the wastewater generated. Responses to the MPP detailed survey indicate that dry cleaning is a much more common practice at meat processing facilities than at 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, which 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 can also reduce water consumption. Continual monitoring of water use and waste loads, however, is a necessity to avoid slippage in performance.

8.7.3 Multiple Use and Reuse of Water

USDA FSIS guidelines do not preclude the multiple use and reuse of water in MPP facilities as practices to reduce potable water consumption and the discharge of treated wastewater. Although 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 meat and poultry processing 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 use in poultry processing is rewarded by savings in processing cost and reduced requirements for wastewater treatment. Examples include using scalder overflow to flume feathers from mechanical de-feathering equipment and using chiller overflow to flume inedible viscera to screens for recovery before rendering. Combination UV irradiation and ozonation can be effective treatment for reused poultry chiller overflow (Diaz and Law, 1997). These are examples of countercurrent recycling, in which water reuse is countercurrent to product flow.

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

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

In the poultry processing industry, a number of unit process-level reuse strategies have also 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 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 that they reuse water from the wastewater treatment plant in the de-feathering 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 the detailed survey responses.

8.7.4 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 wastewaters are recycled for feather fluming.

- Confine bleeding and provide for sufficient bleed time. Recover all collectable blood and transport it to rendering in tanks rather than by dumping it 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 might require preheating the chiller water with the scalder overflow water by using a simple heat exchanger.

- Use pretreated poultry processing wastewaters for condensing all cooking vapors in on-site 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 by-product screening and handling systems. A backup screen might be required to prevent by-products 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 material spills tend to occur.
- 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 wastewaters. 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 wastewater 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 it.

- In-plant primary systems—catch basins, skimming tanks, air flotation, and the like—should provide for at least a 30-minute detention time of the wastewater. 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 considerably below that of Best Practicable Control Technology Currently Available (BPT) except for small processors.
- Reduce the wastewater 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).

8.7.5 Nonregulatory Approaches to Pollution Prevention

EPA is using nonregulatory 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, the American Association of Meat Processors, USDA, several state agencies, EPA programs and regions, and other interested constituent groups. For example, EPA and its partners have developed best management practice guidance materials for the handling and disposal of rendering materials, and for chloride, nitrogen, and phosphorus discharges. The project team evaluated these management practices and developed measures of their effectiveness. The final tools will be deployed over the long term through the active leadership of the industry's trade associations. In addition, EPA partnered 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 provided technical assistance and implementation consulting to the five companies. The pilot was completed in 2002, and EPA evaluated the pilot and incorporated the lessons learned into the final version of the *EMS Guide for Meat and Poultry Processors*. The final guide was completed in summer 2003 and is being marketed throughout the meat and poultry processing industry.

8.8 REFERENCES

Banks, C.J. 1994. Anaerobic Digestion of Solid and High Nitrogen Content Fractions of Slaughterhouse Wastes, In *Environmentally Responsible Food Processing*, AIChE Symposium Series no. 90, 103–109. (DCN 100005)

Banks, C.J., and O.O. Adebowale. 1991. Review of Abattoir By-product Disposal options. In *Proceedings of MLC Conference: Meat Strategies, Options for By-product Disposal*, 26–27th March 1991, Birmingham, U.K., pp. 21–42. (DCN 00042)

Banks, C.J., and Z. Wang. 1999. Development of a two phase anaerobic digester for the treatment of mixed abattoir wastes. *Water Science and Technology* 40 (1). (DCN 10065)

Berthouex, P.M., D.L. Grothman, D.O. Dencker, and Lawrence J.P. Scully. 1977, May. *Characterization and In-Plant Reductions of Wastewater from Hog Slaughtering Operations*. EPA-600/2-77-097. (DCN 10058)

Brooks, D.R., and G. Van Stone. 2001. UV experience in inactivating *Cryptosporidium* in surface water plants. *WaterWorld*, May 2001. PennWell Publishers. (DCN 10068)

Carawan, R.E. 1994. Overview of Water Recycling in Processing. In *Proceedings 1994 National Poultry Waste Management Symposium*, ed. P.H. Patterson and J.P. Blake, pp. 211–215. National Poultry Waste Management Symposium Committee, Auburn University, Alabama. (DCN 00198)

Carawan, R.E., and J.S. Clemens. 1994. Using Renovated Process Water at Hatfield Packing. In *Proceedings 1994 National Poultry Waste Management Symposium*, ed. P.H. Patterson and J.P. Blake, pp. 220–230. National Poultry Waste Management Symposium Committee, Auburn University, Alabama. (DCN 00197)

Clanton, C.J. 1997. *Alternative Waste Management Systems*. Department of Agricultural Engineering, University of Minnesota. <http://www.bae.umn.edu/extens/manure/landapp/wasteman.html>. (DCN 00250)

Cowan, J.A.C., F. Mactavish, C.J. Brouckaert, and E.P. Jacobs. 1992. Membrane treatment strategies for red meat abattoir effluents. *Water Science and Technology* 25 (10). (DCN 10066)

Diaz, M.E., and S.E. Law. 1997. Ultraviolet Photon Enhanced Ozonation for Microbiological Safety in Poultry Processing Water. In *Proceeding 1997 ASAE Annual International Meeting*, 10–14 August, 1997, Minneapolis, Minnesota. (DCN 00037)

Eremektar, G., E. Ubay Çokgör, S. Övez, F. Germirli Babuna, and D. Orhon. 1999. Biological treatability of poultry processing plant effluent—A case study. *Water Science and Technology* 40 (1). (DCN 00082)

Food Manufacturing Coalition for Innovation and Technology Transfer. 2002. *Wastewater Reduction and Recycling in Food Processing Operations*. <http://fpc.unl.edu/fmc/7wastewa.htm>. (DCN 00251)

Gelman, S.R., S.D. Scott, and H. Davis. 1989. Waste Minimization in the Poultry Processing Industry—Process and Water Quality Aspects. Presented at MISSTAP Workshop, Waste *Minimization for Mississippi Industries*, Mississippi State University, 9 November 1989. NIST No. PB95-251385. (DCN 00077)

Glenn, S.L., R.T., Norris, Jr., and J.T. Sommerfield. 1990. Discrete-event simulation in wastewater treatment. *Journal of Environmental Science and Health*, A25 (4). (DCN 00079)

Grady, C.P.L., Jr., and H.C. Lim. 1980. *Biological Wastewater Treatment Theory and Applications*. Marcel Dekker, Inc., New York, New York. (DCN 00248)

Green, T., et al. 1981. Case History of Nitrification of a Rendering-Meat Packing Wastewater, In *Proceedings of the 35th Industrial Waste Conference*, Perdue University, ed. J.M. Bell. (DCN 10006)

Harper, S.R., C.C. Ross, G.E. Valentine, and F.G. Pohland. 1999. Pretreatment of poultry processing wastewater in a pilot-scale anaerobic filter. *Water Science and Technology*, 22 (9). (DCN 00034)

Johns, M.R.1995. Developments in wastewater treatment in the meat processing industry: A review. *Bioresource Technology* 54. (DCN 00128)

Johnston, C.A. 2001. *Comparison of Meat Processing and Domestic Wastewaters*. USEPA Memorandum to File. (DCN 10038)

Kiepper, B. 2001. *A Survey of Wastewater Practices in the Broiler Industry*. The University of Georgia and U.S. Poultry and Egg Association. Presented at WEFTEC 2001. (DCN 00260)

Lo, K.V., and P.H. Liao. 1990. Treatment of poultry processing wastewater using sequencing batch reactors. Department of Bio-Resource Engineering, University of British Columbia, *Canadian Agricultural Engineering*, 32 (2). (DCN 00080)

Loehr, R.C., et. al. 1979. *Land Application of Wastes Volume 1*. (DCN 00165)

Martin, E.J., and E.T. Martin. 1991. *Technologies for Small Water and Wastewater Systems*. Van Nostrand Reinhold, New York, New York. (DCN 00259)

Metcalf and Eddy, Inc. 1991. *Wastewater Engineering—Treatment, Disposal, and Reuse*. 3rd ed. McGraw-Hill Publishing Company, New York, New York. (DCN 00213)

Morris, D., et al., 1998. ASBR Treatment of Beef Slaughterhouse Wastewater. *Proceedings of the Industrial Waste Conference. 52nd (1998): pp. 225 - 236*. (DCN 00191)

Nielsen, V.C. 1996. Treatment and Disposal of Processing Wastes. In *Processing of Poultry*, G.C. Mead. Chapman and Hall Publishing Company, New York, New York. (DCN 00133 - DCN 00137)

Ohio EPA (Ohio Environmental Protection Agency). 1999. *National Pollutant Discharge Elimination System: Tiered Permits*. DSW-0100.016. DWS policy manual. Ohio Environmental Protection Agency, Columbia Ohio. (DCN 00257)

Overcash, M.R. and D. Pal. 1979. *Design of Land Treatment Systems for Industrial Wastes–Theory and Practice*. North Carolina State University, Raleigh, North Carolina. (DCN 00163)

Peavy, H.S., D.R., Rowe, and G. Tchobanoglous. 1986. *Environmental Engineering*. McGraw Hill Publishing Company, New York, New York. (DCN 00261)

Randall, C.W., and P.R. Mitta. 1998. *Preliminary Assessment of the Rocco Farm Foods Wastewater Treatment Plant*, Edinburg, Virginia, for Biological Nutrient Removal. Virginia Polytechnic Institute and State University, Department of Civil and

Environmental Engineering, Blacksburg, Virginia. Submitted to the USEPA Chesapeake Bay Program, Annapolis, Maryland. (DCN 00035)

Randall, C.W., Z. Kisoglu, D. Sen, P. Mitta, and U. Erdal. 1999. Evaluation of Wastewater Treatment Plants for BNR Retrofits Using Advances in Technology. Virginia Polytechnic Institute and State University, Department of Civil and Environmental Engineering, Blacksburg, Virginia. Submitted to the USEPA Chesapeake Bay Program, Annapolis, Maryland. (DCN 00031)

Reddy, M., ed. 1998. *Biological and Chemical Systems for Nutrient Removal*. Water Environment Federation, Alexandria, Virginia. (DCN 00253)

Reynolds, T.D. 1982. *Unit Operations and Processes in Environmental Engineering*. PWS-Kent Publishing Company, Boston, Massachusetts. (DCN 00256)

Ross, C.C., and G.E. Valentine. 1992. Anaerobic Treatment of Poultry Processing Wastewaters. In *Proceedings of 1992 National Poultry Waste Management Symposium*. (DCN 00254)

Scaief, J.F. 1975, June. *Effluent Variability in the Meat-Packing and Poultry Processing Industries.* PB-245-623. Pacific Northwest Environmental Research Laboratory, Corvallis, Oregon. (DCN 10001)

Sen, D., C.W. Randall, and T. J. Grizzard. 1990. *Biological Nitrogen and Phosphorus Removal in Oxidation Ditch and High Nitrate Recycle Systems*. Virginia Polytechnic Institute and State University, Department of Civil Engineering, Manassas, Virginia. (DCN 00029)

Starkey, J.E., and T. Wright. 1997. Tertiary Screening in Poultry Wastewater Treatment. Presented at *1997 Poultry and Environment Management Seminar*. (DCN 10074)

Stebor, T.W., C.L. Berndt, and R. Gabriel. 1990. Operating Experience: Anaerobic Treatment at Packerland Packing. In *44th Purdue Industrial Conference Proceedings*. Lewis Publishers, Inc., Chelsea, Michigan. (DCN 00262)

Uhlman, K. 2001. Land Application for Natural Wastewater Treatment. *Environmental Protection*, 12 (8, August). (DCN 10069)

USEPA (U.S. Environmental Protection Agency). 1974, February. *Development Document for Effluent Limitation Guidelines and New Source Performance Standards for the Red Meat Processing Segment of the Meat Product and Rendering Processing Point Source Category*. U.S. Environmental Protection Agency, Washington, D.C. (DCN 00162)

USEPA (U.S. Environmental Protection Agency). 1975, April. *Development Document for Effluent Limitation Guidelines and New Source Performance Standards for the Poultry Segment of the Meat Product and Rendering Processing Point Source Category*. U.S. Environmental Protection Agency, Washington, D.C. (DCN 00140)

USEPA (U.S. Environmental Protection Agency). 1980. *Treatability Manual*. EPA/600/8-80/042d. U.S. Environmental Protection Agency, Washington, D.C.

USEPA (U.S. Environmental Protection Agency). 1987. *Design Manual - Phosphorous Removal*. EPA/625/1-87/001. U.S. Environmental Protection Agency, Washington, D.C. (DCN 00255)

USEPA (U.S. Environmental Protection Agency). 1993. *Manual - Nitrogen Control*. EPA/625/R-93/010. U.S. Environmental Protection Agency, Washington, D.C. (DCN 10023)

USEPA (U.S. Environmental Protection Agency). 1997, September. *Estimates of Global Greenhouse Gas Emissions from Industrial and Domestic Wastewater Treatment*. EPA-600/R-97-091. U.S. Environmental Protection Agency, Washington, D.C. (DCN 10061)

Weber, W.J., Jr. 1972. *Physicochemical Processes for Water Quality Control*. John Wiley & Sons, New York, New York. (DCN 00252)

Witherow, J.L., A. Hamza, and S. Saad. 1978. Water Reuse in Poultry Processing, Prepared for 1978 Summer Meeting of American Society of Agricultural Engineers, 27–30 June, 1978. ASAE Technical Paper No. 78-6026, NIST No. PB-283-695. (DCN 10060)

Zhang, R. 2001. *Biology and Engineering of Animal Wastewater Lagoons*. University of California, Davis. (DCN 00258)