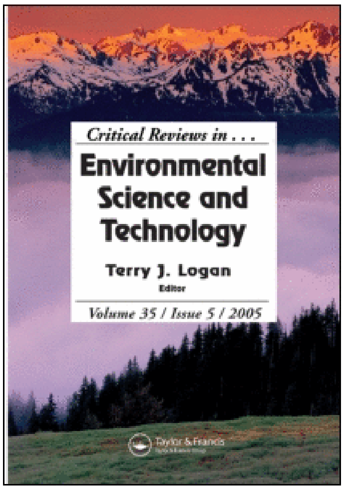


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# Sustainable Natural Systems for Treatment and Disposal of Food Processing Wastewater

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*This paper presents a state-of-the-art review of natural wastewater treatment technologies selected from the point of view of sustainability and relates them to feasible reuse and disposal practices available for food processing wastewater. Selected technologies include land application, constructed wetlands, and various pond systems that all make use of natural processes. The aim of the review is to help understand issues controlling wastewater reuse and how the different natural treatment systems and their combinations could help us to protect the environment, meet regulations, and conserve water, material, and energy resources.*

**KEY WORDS:** constructed wetland, food processing, land application, treatment pond, solar evaporation, sustainable technology, wastewater treatment

## INTRODUCTION

The sustainability of human activities (predominantly production and consumption) is a growing concern among businesses, customers, governments, international bodies, and non-governmental organizations (Graedel & Klee, 2002). These concerns are often linked to energy efficiency, reduction of environmentally harmful emissions, ecosystem preservation and other “save the Earth” efforts. Despite its increasing importance, current definitions of “sustainability” are somewhat vague. The most commonly accepted description

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was provided by the World Commission on Environment and Development (1987) in the so-called Brundtland Report. According to this report, the goal of sustainability is to "meet the needs of the present generation without compromising the ability of future generations to meet their own needs" (p. 24). Other descriptions are similarly phrased and often confuse sustainability with environmental protection and other lofty goals that, strictly speaking, are not required for sustainable operations.

Applications of sustainability framework to wastewater treatment and management are very recent and limited (Jeppsson & Hellstrom, 2002; Levine & Asano, 2004; Maurer et al., 2003; Raluy et al., 2005; Tangsubkul et al., 2005). A somewhat larger number of cases of water resources management have been studied from the sustainability perspective (Bastian, 2005; Langergraber & Muellegger, 2005; Lienert et al., 2006; Loucks et al., 2000; Lundie & Peters, 2005; Lundie et al., 2004, 2005; Miller, 2006). In some sense, sustainability is not a completely new issue. In water resources management, sustainability of water resources, both surface and groundwater, have been recognized for a long time. Originally, sustainability simply meant meeting human demands by natural supplies. As the demand for human consumption, agriculture, and later industry grew, the most easily reachable resources became insufficient. The solution was to find another source further out and bring water where it was needed, again to balance supply and demand. In this approach, sustainability was primarily seen as a technical and managerial issue. Other elements were gradually introduced, and in 1998 the ASCE Task Committee for Sustainability Criteria proposed the following definition: "Sustainable water resource systems are those designed and managed to fully contribute to the objectives of society, now and in the future, while maintaining their ecological, environmental, and hydrological integrity" (ASCE Task Committee for Sustainability Criteria, 1998, p. 44).

Despite this and other efforts, there are no clear criteria of sustainability that could be applied in practice to assess different waste management solutions, especially at technical levels. While the development of such criteria or even a more general discussion of sustainability principles is not the goal of this paper, we decided (quite arbitrarily) to limit the scope of our presentation to selected techniques for wastewater management in the food processing industry. The selection is based on the amount of exogenous energy required for each process, as such energy must currently be provided mostly from non-renewable sources such as fossil fuels. Thus, we decided to exclude from our discussion these treatment processes and systems that require major energy inputs. Activated sludge and membrane filtration are two examples of processes that were not included, as they consume substantial quantities of high-quality energy (primarily electrical) although both processes can be used for treating food processing wastewater. In contrast, processes such as land application or solar evaporation rely mostly on low-quality energy. Biological processes in soil extract energy from wastewater itself by transforming reduced organic and inorganic compounds to higher oxidation states, unlike natural evaporation, which utilizes solar energy directly. The use of exogenous energy in such processes is limited and primarily used for transporting (pumping) the process stream and for ancillary operations. While energy itself does not disappear, its quality, defined as the ability to perform work in a particular environment, does change. This consideration led to a rigorous definition of exergy as a quantity of useful work (Ayres, 1994; Szargut et al., 1988), although applications to wastewater treatment are limited (Hellstrom, 1997; Wilsenach et al., 2003).

We believe that the choice of processes included in this paper is also relevant from another perspective. Although "high-tech" treatment processes (such as engineered bioprocesses) can perform well (and indeed have been employed for food wastewater treatment), they often require higher capital and operational costs and more skilled operators (Bixio & Wintgens, 2006; Liu, 2007). Many food processing operations are relatively small, both in the scale of production and economic profits. These facilities are often located in rural areas where sufficient land is available (unlike in larger cities). Thus, these food-processing plants may be ill-suited for more sophisticated and complex processes and are leaning toward simpler treatment techniques. Such simpler solutions may (if properly designed, maintained, and operated) provide equally good environmental protection over a long period of time while utilizing mostly renewable sources of energy.

This review of natural wastewater treatment and disposal technologies can be used to carry out a preliminary technology feasibility assessment for a certain type of wastewater. In a wider context, the review is mainly targeted for managers, decision-makers, and scientists working in the fields of food processing, environmental and agricultural issues, and regional planning, who don't necessarily have in-depth knowledge of wastewater treatment processes but are looking for a source of proven experience of the performance of these technologies or are interested in sustainable technologies in general.



## LAND APPLICATION

### Design and Selection Criteria

Land discharge (land application or land treatment of wastewater) is widely used for the treatment and disposal of biosolids and wastewater from food processing industries, because of the rural location of many food processing facilities and suitability of food processing waste for land application (Chrobak, 2002; Crites et al., 2006; Papachristou & Lafazanis, 1997; United States Environmental Protection Agency [USEPA], 2006; Walsdorff et al., 2005). However, land application can only be considered as a treatment practice if it is used under conditions that are favorable to natural treatment processes.

Land application is sometimes considered to be the most economical way (and thus the only way, presumably) to meet the zero discharge goal of the U.S. Clean Water Act of 1972 (Crites et al., 2000), which aimed at eliminating the discharge of pollutants into navigable waters by 1985 (United States Department of Justice, n.d.). This goal can be met by evaporation, closing the loops, or treating wastewater so that it can be safely disposed or reused. Food processors are equally committed to pollution prevention. The wastewater treatment goal of environmentally friendly food processing is viewed as providing maximum recovery and reuse of the polluting substances while considering the cost of utilization and profits such as market and fuel value of the recovered substances (Hansen & Hwang, 2003; Song & Hwang, 2003).

Considerable information is available from field application sites to provide a basis for the design of sustainable land application systems (Crites et al., 2006; USEPA, 2006). Land application systems are perceived as low-technology options that do not require complicated engineering structures and continuous process control. However, careful characterization and monitoring of wastewater parameters and the application site is necessary to establish and maintain the desired functions in an environmentally sustainable manner. The purpose of wastewater treatment and water reuse (such as irrigation or aquifer replenishment) has a major impact on site selection and design criteria.



Wastewater treatment systems are classified as primary (mechanical), secondary (conventionally activated sludge process), and advanced (tertiary, polishing) treatment. Land application systems are typically designed to provide secondary treatment or advanced water treatment (polishing) for pretreated wastewater (Crites et al., 2006). When appropriate hydraulic loading rates are used, land application processes typically reduce wastewater BOD, total suspended solids (TSS), and total N levels to 10 mg/L or less, making these technologies suitable for all reuse options specified by USEPA (2004). However, conformation to the requirements should be demonstrated for the specific application site and wastewater type. An adaptation period is needed to achieve optimal biological treatment capacity.

Common pretreatment practices include screening and primary treatment to remove coarse solids, and to reduce organic matter content (Paranychianakis et al., 2006). Pretreatment of food processing wastewater may require pH adjustment and removal of fats and oils by air flotation (Bustamante et al., 2005; Crites et al., 2006; Paranychianakis et al., 2006). As a good example of a complete treatment sequence for dairy processing wastewater, the Australian EPA (1997) recommends a sequence of segregation, screening, equalization and pH control, fat removal, reduction of biological oxygen demand (BOD or $BOD_5$) by physical or biological techniques,

**TABLE 1.** Typical design parameters for natural treatment systems

| | Area requirement (ha(dm$^3$/s)$^{-1}$) | Typical area (ha) | Hydraulic loading | Typical BOD loading (kg ha$^{-1}$d$^{-1}$) | Detention time (d) |
|---|---|---|---|---|---|
| Land application | | | | | |
| Slow rate (SR) | 0.55–5.2 | 23–280 | 0.5–6.0 m/yr | 110–330 | |
| Soil aquifer treatment (SAT) | 0.03–0.55 | 3–23 | 6.0–125 m/yr | 112–667 | |
| Overland flow (OF) | 0.15–1.0 | 6–40 | 3.0–20 m/yr | 5–100 | |
| Constructed wetland | | | | | |
| Free water surface (FWS) | | 1.0* | <189 m$^3$/d† | 200, 45–60‡ | 7–15 |
| Vegetated submerged bed (VSB) | | 0.5* | <189 m$^3$/d† | 600, 60‡ | 3–14 |
| Stabilization pond | | | | | |
| Oxidation pond | | | | 40–120 | 10–40 |
| Facultative pond | | | | 22–67 | 25–180 |

*Median size of 138 FWS and 49 VSB wetlands.
†Mostly used loading for municipal wastewater in the United States.
‡Recommended for municipal wastewater for secondary treatment goal.
References: Crites et al., 2006; Interstate Technology and Regulatory Council (ITRC), 2003; USEPA, 2000



and land irrigation as the best treatment practice. Wastewater streams that could be segregated from the main wastewater stream include clean storm water, reusable components such as whey and spent cleaning solutions, and highly saline wastewater that could be evaporated for salt recovery. Depending on wastewater characteristics and the treatment efficiency of the land application system, less pretreatment may be needed.

Another pretreatment concept based on physical-chemical and biological processes has been presented by Chrobak and Ryder (2005) and specifically designed for distillery process water. After screening and flow equalization, the effluent is treated in anaerobic bioreactors, re-aerated, and adjusted to a higher pH. Biogas and heat formed in the process are recovered.

Land application designs have been classified as slow rate systems (SR), soil-aquifer treatment (SAT) (or rapid infiltration), and overland flow (OF). Each category has specific requirements and capacities that should be taken into account when selecting and designing sustainable treatment technologies for food processing wastewater (see Table 1). Essentially, either the application method or site must be chosen so that it conforms to the wastewater composition and production rate. Decisive site characteristics include the size of the available land area, soil permeability, removal or detention capacity of organics and nutrients, and vulnerability of the surrounding environment (Liu, 2007). For more detailed information, the cited design handbooks and local regulations should be reviewed. Performance data and expected effluent quality for properly designed and maintained land application systems are shown in Table 2.

**TABLE 2.** Observed and expected (based on design models) treatment capacities of natural treatment systems

| | BOD | | | TSS | | | Total N | | | Total P | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Influent (mg/L) | Effluent (expected) (mg/L) | Removal (%) | Influent (mg/L) | Effluent (expected) (mg/L) | Removal (%) | Influent (mg/L) | Effluent (expected) (mg/L) | Removal (%) | Influent (mg/L) | Effluent (expected) (mg/L) | Removal (%) |
| Land application | | | | | | | | | | | | |
| SR | — | — (<2) | — | 30–274* | <1–80* | 96 . . . > | 7–66† | 2–20† | 50–93† | — | (<0.1) | — |
| SAT | 15–228 | 0–58 (5) | 75–100 | | (SR <2, SAT 2) | 99* | 12–50† | 2.8–20† | 38–93† | 2.1–11 | <1–7.4 (<1) | 29–99 |
| OF | 18–507 | 5–23 (10) | 46–98 | — | — (10) | — | | (SR 3, SAT 10, OF <10) | | — | — | 40–50 |
| Constructed wetland | | | | | | | | | | | | |
| FWS | 26–140 | 5.4–19 | 54–88 | 9.2–380 | 4.8–53 | 23–93 | — | — | 33–45/85§ | 1.9–11.1 | 0.1–5.7 | 10–98 |
| | 113‡ | 22 | 81 | 112‡ | 20 (5–15) | 82 | 14§§ | 8.4 (5–10) | 41 | 4.2§§ | 2.2 | 49 |
| VSB | 23–118 | 1.7–25 | 65–88 | — | — | —# | 20–25 | — | 20–70 | 7–10** | — | 10–40** |
| | 50–350‖ | — | 85 | 100–1,000‖ | — | 91 | 150–250‖ | — | 25 | | | 41 |
| | | (5–10) | | | (5–20) | | 47§§ | 27 (5–10) | 42 | 8.8§§ | 5.2 | |
| Stabilization pond | | | | | | | | | | | | |
| Oxidation | — | (20–40) | — | — | (80–140) | — | — | — | — | — | — | — |
| Facultative | — | 50–70†† (30–40) | 75–85†† | — | (40–100) | — | — | — | 20–80‡‡ | — | — | 20–50‡‡ |

*SR and SAT.
†All land application systems (not specified).
‡Mean concentration of primary treated municipal wastewater, n = 22.
§Without/with pre-nitrification (Seabloom & Hanson, 2005; Wass, 2006).
‖Lyon, 2006. Dairy wash water, with average removal.
#Similar to FWS wetlands.
**Typical influent level and expected performance.
††Without filtration (Walmsley & Shilton, 2005).
‡‡Craggs (2005a).
§§Vymazal (2007).
References: Crites et al. (2000, 2006), unless otherwise indicated.

## Slow Rate (SR) Systems

SR systems are typically facilities constructed for spray irrigation of, for example, pastures, forests, and golf courses, in which wastewater purification takes place on soil surface, in vegetation layer, and within the soil matrix. Percolating water is not collected but it can be used to replenish an underlying groundwater aquifer. The surface runoff of applied wastewater is known as tailwater and must be contained on-site (Crites et al., 2006). SR systems can be designed either to treat a maximum volume of wastewater in the soil-aquifer system (slow infiltration–type process) or to optimize the irrigation potential by reusing a maximum amount of nutrients as fertilizers at the site (crop irrigation–type process). SR systems can be loaded only with lower volumes of wastewater and lower concentrations of organic matter (expressed as BOD) and nutrients compared to SAT systems (see Tables 1 and 2). However, these relatively low hydraulic loading rates and long travel distances are generally used to provide quality with better effluent than in the other treatments (Crites et al., 2006; USEPA, 2006). Soil for SR systems should be fine enough to allow sufficient residence time. Thus, sandy loams to clay loams with permeability between 0.15 and 15 cm/h are preferred (Crites et al., 2006). It is also important to maintain high microbial activity in soil. Freezing of soil reduces infiltration rates, and thus, the recommended lowest mean temperatures for land application range between 0°C and –4°C (USEPA, 2006).



Vegetation at the site may affect dramatically the performance of SR systems through its effects on permissible hydraulic loading, nutrient uptake, biomass production, structure and activity of microbial community, and the fate of trace elements and organic contaminants (Paranychianakis et al., 2006). SR systems can be designed to produce economically valuable crops. In addition, harvesting of crops potentially can remove some of the nutrients and salts. However, there is not much evidence on the extent to which the different salinity components are actually removed and utilized. SR treatment of milk-processing wastewater has shown hardly any reduction in salinity of the percolate as compared to the applied wastewater (Crites et al., 2000).

Due to low hydraulic loadings, evaporation and plant uptake consume most of the wastewater, so that there typically is hardly any possibility to replenish groundwater or surface water supplies. If an SR system involves irrigation of crops, influent wastewater should meet special qualifications (USEPA, 2004). To maintain good plant growth, it may be necessary to adjust extreme pH conditions in waste-water (USEPA, 2006) to values between 5 and 9. If the SR system is not used to irrigate crops, pretreatment of wastewater may not be necessary. In general, it is recommended to restrict irrigation with wastewater to areas with groundwater table lower than 1.5–3 m and the distance of 500–1000 m from surface water bodies (World Health Organization [WHO], 2006).

### Soil Aquifer Treatment (SAT) Systems

The systems of SAT, defined as "a three-component treatment process consisting of the infiltration zone, vadose zone, and aquifer storage" (Fox et al., 2006, p. xxv), differ from those of SR mainly by the higher rate of wastewater application. They can be subjected to the highest BOD loading among the land application designs, yet the expected effluent BOD concentration of approximately 5 mg/L is considered acceptable for various reuse options (Salgot et al., 2006; State Water Resources Control Board, 2005; USEPA, 2004; WHO, 2006). Hydraulic loading rates are also high, and the influent quality and site characteristics must be such that percolation of salts, nitrates, or other wastewater components does not impair the beneficial use of groundwater or further land use. Thus, site requirements are more stringent than for the other land application systems. Furthermore, clogging of soil may require lower loading rates, and the entire lifecycle (duration of operation) of SAT systems is likely to be shorter than those of SR systems. SAT effluent can be used to recharge groundwater aquifer or collected for other reuse.

Removal of phosphates improves with increasing travel distance in soil and the treatment time, whereas nitrates and salts are more likely to travel over extended distances with much less attenuation. Denitrification in soil is the main mechanism for nitrogen removal, whereas crop uptake and evaporation have much smaller contributions. It is possible that denitrification may be limited by the quantity of available organic carbon. The ratio of carbon to nitrogen (C/N) depends on the nature of organics. For acetate, a theoretical minimum C/N ration is almost 1 g C/g N, but in real applications, carbon requirements are likely to be higher due to growth requirements and microbial competition, pH should be higher than 5.5, and temperature should be at least 0°C but preferably >5°C to support nitrogen removal. The freezing of soil and the need to create aerobic/anaerobic cycles for nitrogen removal and prevention of ammonium breakthrough imply that SAT sites cannot be operated continuously (Bixio & Wintgens, 2006), which does not necessarily hamper food processors with seasonal or cyclic wastewater production. At eight U.S. sites, the ratio of the total duration of wet to dry periods has been 0.03–0.75 (USEPA, 2006). To accommodate these cyclic wastewater applications, temporary storage tanks or ponds for wastewater storage may be needed, and vegetation should be selected to tolerate wet conditions.

### Overland Flow (OF) Systems

Wastewater treatment in OF systems occurs as wastewater flows down vegetated slopes and becomes purified by physical, chemical, and biological processes occurring on the soil surface and vegetation cover. As percolation of wastewater into deeper soil layers is limited, there is a stream of treated effluent to be captured at the bottom of the sloped field.

OF systems, unlike SR and SAT, are best suited for low-permeability soils. Surface waters and atmosphere are the major sinks of applied water and pollutants. Proprietary runoff collection systems can be used to capture the treated effluent and direct it to a final disposal site. Topography of the OF site can be slightly sloping. Slope lengths in OF practice have typically ranged from 30 to 60 m. The longer the slope the greater the removal of BOD, TSS, and nitrogen (USEPA, 2006). Applicable hydraulic loading rate is higher and area requirement smaller than in SR systems.

OF treatment is suited for warmer seasons when it can be used nearly continuously, except for possible drying periods before cutting the grass. Removal processes mainly occur on soil surface, and the treatment and storage capacity of subsurface soil cannot be utilized. Removal of high BOD concentrations (about 800 g/L) is limited by the oxygen transfer efficiency (USEPA, 2006). Experience with food processing wastewater has shown that BOD loading can mostly be around 110 kg $ha^{-1}d^{-1}$ without impairing removal (Crites et al., 2006). Thick grass cover and slow flow velocity are favorable for TSS removal (USEPA, 2006). Like the SR systems, vegetation plays a major role in wastewater treatment. Phosphorus removal in OF systems is limited to about 40–50% (Crites et al., 2006). However, OF is well-suited for nitrogen removal. Experience from tomato processing has shown no removal of salinity components during land treatment, but dilution and precipitation of salts and minerals in rivers and streams makes them feasible recipients for low-salinity wastewater effluents (Crites et al., 2000).



## Advanced Land Application Systems

Strategies that might improve the feasibility of land application systems include sequential reuse and integration of land application with wetland treatment or solar evaporation. Sequential reuse and integrated on-farm drainage management (IFDM), which involves final disposal of brine to a solar evaporator, has been developed primarily for the management of salinity in agricultural drainage. However, they might be considered also for relatively low-salinity (<8 dS/m or <6700 mg/L TDS) and low-strength wastewater from food processing industries. These technologies are better suited for relatively small on-site systems rather than for regional-scale salinity management. In addition to saline wastewater, they require a supply of freshwater for blending or cyclic use during certain portions of the growing season. Gypsum amendments can also be used to mitigate adverse impacts of sodium. Sequential reuse and IFDM have been successful in field tests, and extensive research and development is still ongoing. The main benefits include the ability to produce higher value crops, manage salinity, and maintain groundwater levels in regions where salinity impairs farming (Cismowski et al., 2006; Murtaza et al., 2006; State Water Resources Control Board, 2004).

## Environmental Impacts and Sustainability

### Long-Term Effects

Sustainability of land application systems has been examined in several projects, suggesting that subsurface processes have a high capability of treating metals and organic compounds and land application can be sustainably operated under a wide range of conditions (Bastian, 2005; O'Connor et al., 2005; Overcash et al., 2005). On the other hand, the fate of all chemical compounds has not been explored; for example, polyphenols and lignins that are abundant in fruit and vegetable processing wastewater are known to be slowly biodegradable (Grismer et al., 2003).

Experiments with pretreated food processing wastewater or sludge have not shown detrimental effects of long-term land application on soil sodicity, salinity, nutrients, organic carbon, or soil aggregation (Allinson et al., 2007; Virto et al., 2006). However, Cruz et al. (1991) reported on elevated nitrogen concentrations in groundwater during a 15-year application of vinasse. Long-term experience (8–31 years) from the land application of municipal waste biosolids has also demonstrated occasional increase in nitrogen and heavy metal levels in soil, groundwater, and surface water runoff (Schroder et al., 2008; Surampalli et al., 2008; Tian et al., 2006). However, some properly maintained land treatment systems for municipal wastewater have been in use for more than 100 years (Crites et al., 2000).



### Ecosystem Impacts

Land application affects the structure and functions of the ambient ecosystem by wetting and clogging the soil and by adding nutrients, organic matter, and salts. Land application at continuously high hydraulic loading rates eventually leads to the extinction of plant species that are not adapted to oxygen-deficient, saturated habitats (Paranychianakis et al., 2006). For the purposes of agricultural irrigation, the hydraulic loading rates for SR systems, 0.5–6 m/yr, are typically compatible with agricultural irrigation needs (0.93–5.6 m/yr) (Crites et al., 2006). However, higher hydraulic loading rates in SR and SAT systems eventually leads to the mixing of wastewater effluent with groundwater.

High hydraulic loading rates accompanied by reduced infiltration might also increase the risk of soil erosion. The counteracting impacts of wastewater salinity and organic matter on soil infiltration are yet to be discussed in this report.

### Impacts of Organic Matter and Nutrient Loading

An important benefit of land application systems is the possibility to recycle organic matter and nutrients on fields and pastures. Recycling of phosphorus reduces the environmental impacts of phosphate mining and eutrophication and saves limited phosphate reserves (WHO, 2006). In general, application of organic matter and nutrients on land is sustainable under loading rates that can be assimilated by plants.

BOD concentrations of 110–400 mg/L have beneficial impacts on soil and crops, as indicated by increases in microbial activity, soil fertility, and productivity (WHO, 2006). Sorption on organic matter (natural and derived from waste) reduces the mobility and bioavailability of trace metal salts present in wastewater (Basta et al., 2005). On the other hand, excess organic loading can result in malodorous anaerobic conditions, leakage of organic matter and mobilized metals, and increase in alkalinity (USEPA, 2006). At BOD loadings higher than 300 kg $ha^{-1}d^{-1}$ and TSS loading higher than 112 to 224 kg $ha^{-1}d^{-1}$, a more careful management of the system is needed to avoid problems with odors and soil clogging (Crites et al., 2006; Paranychianakis et al., 2006). In practice, BOD loading rates that have been used for land application of food processing wastewaters at 11 sites in the United States have ranged from 84 to 448 kg $ha^{-1}d^{-1}$ (Crites et al., 2006). In Australia, BOD concentration less than 60 mg/L is recommended for irrigation to control odor formation (Australian Environmental Protection Agency, 1997), which shows that odors can be restricted via the beneficial use of BOD components in sensitive areas.

To provide sufficient removal of nitrates, wastewater should contain an appropriate carbon to nitrogen ratio to promote denitrification and anaerobic conditions (WHO, 2006). Leaching and surface runoff of nitrates and ammonia may lead to odor problems and toxic effects on human (infants in particular) and animal health, and, together with phosphates, contribute to algal blooms and production of cyanobacterial and algal toxins (Pierzynski & Gehl, 2005; WHO, 2006). Much research has been done to estimate sustainable nutrient loading rates that allow wastewater reuse for irrigation and replenishment of aquifers without nitrogen accumulation in the receiving soil-aquifer system (Crites et al., 2006; McCardell et al., 2005; Pierzynski & Gehl, 2005). Sustainable nitrogen loading depends strongly on site-specific removal and transportation characteristics; thus, the use of general guidelines can be questionable, and local studies should be conducted. For example, total denitrification losses measured at wastewater application sites are in the range of 200–250 kg $ha^{-1}$ $year^{-1}$, and this application rate is recommended for irrigation sites in Australia (Australian Environmental Protection Agency, 1997; New Zealand Institute of Chemistry, n.d.). This implies that with an influent nitrogen concentration of 50 mg/L, hydraulic loading should not exceed 0.5 m $ha^{-1}$ $year^{-1}$, which is congruent with the typical design of SR systems. If volatilization of ammonia is significant and crops with high nitrogen uptake are harvested, higher loading may be sustainable. Plant uptake rates of up to 670 kg $ha^{-1}$ $year^{-1}$ of nitrogen have been measured (Crites et al., 2000).

Plant uptake rate of phosphorus is limited to about 84 kg $ha^{-1}$ $year^{-1}$ (Crites et al., 2000). However, application of phosphorus on SR and SAT sites entails lower risks of groundwater contamination than nitrogen application. Soil provides storage for phosphorus, especially if it contains clay minerals. Experience from irrigation with dairy factory effluent has shown that after applying a total of 12.6 t/ha (573 kg $ha^{-1}$ $year^{-1}$) of phosphorus over 22 years, approximately 91% of it was found to be within a 0–0.75 m soil layer, and it was mostly in a form that is available to plants. In contrast, only 8% (76 kg $ha^{-1}$ $year^{-1}$) of the total nitrogen load of 21 t/ha was stored in soil, mostly in the 0.1–0.5 m layers (Degens et al., 2000).

Year-round production of wastewater, like in potato processing, may require alternative treatment and disposal methods. When the ground is frozen, application of wastewater with even low phosphorus content may accelerate leaching (Zvomuya et al., 2006).

## Salinity

Some salinity components of food processing wastewater can also be reused as micronutrients. However, salinity is one of the critical factors impairing sustainable land application of certain food processing wastewaters. Leaching of salts to groundwater may occur over time after cessation of salt application (Thunqvist et al., 2004). Tolerance of crops to salinity varies by species. Sorghum seems to be the most sensitive of the common crop plants, with a 10% yield reduction at electric conductivity of 5.9 dS/m. Field and forage crops are in general more tolerant to salts than vegetable crops (USEPA, 2006). Salinity inhibits nutrient uptake and biomass production and hence nutrient removal to a greater extent than water consumption (Paranychianakis et al., 2006). To maintain good environmental quality, land application rates should be adjusted to both lowered water consumption, nutrient removal, and BOD mineralization (Nelson et al., 1996; Shani et al., 2005). Sufficient leaching is needed to prevent salt accumulation in the root zone, but this practice may contribute to problems with agricultural drainage disposal.

The usability of groundwater as drinking water is threatened at total dissolved solids (TDS) content >500 mg/L. To prevent clogging and corrosion of sprinklers, wastewater should contain less than 100 mg Cl/L, 70 mg Na/L, and <1.5 mg Fe/L or Mn/L (WHO, 2006). Swelling of clay minerals and deflocculation of soil colloids, and the resulting impairment of soil infiltration, is more difficult to control by giving loading recommendations. Even though these impacts are mainly caused by sodium ions, the magnitude of damage is co-influenced by other wastewater components, such as carbonates and bicarbonates, as well as characteristics of the application site. Recent review shows that organic matter may either improve or impair soil infiltration (Paranychianakis et al., 2006). Stability of soil aggregates, and therefore soil infiltration, was found to increase after long-term irrigation with dairy factory effluent, an effect attributed to the presence of lactose in the effluents (Cameron et al., 2003). This might counteract the impacts of high sodium adsorption ratio (SAR) and salinity found in some food processing wastewaters (Crites et al., 2000; Virto et al., 2006).

Due to the complex interactions and different ways to measure salinity, sustainable loading rates have been approximated using combinations of parameters. When soil is irrigated with wastewater with TDS 250–850 mg/L, conductivity <3 dS/m, SAR 5–9, and sodium <100 mg/L, no observable short-term effects are expected, while the long-term soil salinization depends on leaching and drainage properties. However, problems in sensitive crops may occur with TDS 450–2000 mg/L and conductivities from 0.7 to 3 dS/m. The limit that prevents all sustainable use is probably at wastewater concentrations with TDS 2000 mg/L, conductivity 3 dS/m, SAR 8, and sodium 1000 mg/L, above which the soil structure and capacity to sustain plants is lost (WHO, 2006). Soils with low content of nonswelling clays can tolerate higher SARs (Crites et al., 2000). In Australia, it is recommended to use wastewater with less than 1000 mg/L of TDS to irrigate pastures (Australian Environmental Protection Agency, 1997).



#### Fats and Oils

Fats, oils, and grease are often present in wastewater of both animal and vegetable origin. Fats and oils may decrease gas exchange between soil and air and impair percolation of water within about 2–8 weeks of application. Seed germination is also affected. Fats and oils alter the nitrogen cycle, and, hence, the growth of plants and bacteria is impaired. Reduction in crop yield has been observed at fat and oil doses of 1–2% of soil weight. However, wastewater application may restore the nutrient balance and accelerate oil biodegradation. In the long term, as the hydrocarbons are decomposed, the growth of plants is often improved because of increased soil fertility and improved physical conditions (e.g., moisture retention) (Overcash & Pal, 1979).

## CONSTRUCTED WETLANDS

### Design and Selection Criteria

Constructed wetlands, or treatment wetlands, are artificial wastewater treatment systems that employ the same biological processes found in larger natural wetland ecosystems but are more capable of tackling the problems with high seasonal and spatial variation in treatment efficiencies that are associated with natural wetlands (Seabloom & Hanson, 2005; Wetzel, 2001). Compared with wastewater treatment ponds, constructed wetlands have higher hydraulic efficiency and lower effluent TSS than ponds (Kadlec, 2005). Constructed wetlands have been used for wastewater treatment since the early 1950s (de Feo et al., 2005). For the treatment of municipal wastewater and stormwater, these technologies are mature, tested, and now being used in new applications and, in some cases, on new contaminants (ITRC, 2003).

Several guidance documents have been published to assist the design of constructed wetlands so that the overall treatment objectives can be achieved (USEPA, 2004; Wallace, 2005). However, constructed wetland technologies are considered to be still in their evolutionary stage, because the internal biotic and abiotic processes occurring in wetlands have not been adequately quantified. In particular, it has been difficult to model the reduction of externally applied versus internally produced TSS and BOD loads (Wallace, 2005).

In general, constructed wetlands are considered to be suitable for secondary and tertiary treatment of municipal wastewater but only for tertiary treatment of industrial wastewater (ITRC, 2003; Sustainable Conservation, 2005). However, some types of food processing wastewater, such as wastewater from fruit and vegetable processing and wineries, do not contain significant levels of toxins and metals or high levels of organic matter and nitrogen. After primary treatment to remove most suspended solids, these types of food processing wastewater are considered to be suitable to create a healthy wetland habitat (O'Brien et al., 2002). Experience from the treatment of industrial wastewater shows that wetland systems are also capable of removing 54–94% of oil and grease (ITRC, 2003). With regard to other food processing residuals, dissolved salts can be problematic, as they are not removed unless wetland vegetation is harvested (Sustainable Conservation, 2005).

Site requirements for wetlands are not as stringent as for land application sites (Crites et al., 2006). Constructed wetlands can be used in areas with high water table and/or low permeable soil (USEPA, 2004). They are particularly suited for sites located adjacent to restorable wetlands and/or upland areas needed to provide adequate buffer, or on previously drained wetlands (Sustainable Conservation, 2005). Cold temperatures impair the treatment efficiency. High evapotranspiration in arid climates leads to loss of water resources and increased effluent salinity (ITRC, 2003; Wass, 2006).

Constructed wetlands are not recommended for short duration time-critical cleanups. It may take one to three years to establish feasible biological functioning of an artificial ecosystem, and required residence times are long (ITRC, 2003; O'Brien et al., 2002; Wetzel, 2001). On the other hand, constructed wetlands have low requirements for operation and management (de Feo et al., 2005).

Experience collected from California food processors in 2002 (O'Brien et al., 2002) showed that occasional control of nuisance weeds, rodents, and mosquitoes is needed, and large numbers of waterfowl can cause effluent from wetlands to contain significant levels of fecal coliform. The effect of shock loadings from food processing on wetlands had been minimal. However, some food processors were concerned about the lacking evidence of feasibility for specific types of wastewater and the long-term effectiveness of the systems. More recently, full-scale feasibility studies have been carried out with winery, dairy, and seafood processing effluent (Grismer et al., 2003; Muñoz et al., 2006; Yirong & Puetpaiboon, 2004).

## Free Water Surface (FWS) Wetlands

Constructed wetlands can be classified based on the type of vegetation and flow regime (Vymazal, 2001; Wallace, 2005; WHO, 2006). Free water surface (FWS) or surface-flow wetlands are artificial ecosystems that mimic natural marches and swamps and consist of one or more shallow basins (see Figure 1). At least some of the water surface is freely aerated by atmospheric oxygen, but in densely vegetated zones, aeration and algal growth can be limited. Oxygen is also scarce near the bottom of the wetland.

Both FWS and VSB constructed wetlands have been used in municipal wastewater treatment to meet a 30 mg/L BOD and 30 mg/L TSS secondary discharge standard (ITRC, 2003). For FWS, these standards can be met with maximum monthly influent rates of 60 kg $ha^{-1}d^{-1}$ for BOD and 50 kg $ha^{-1}d^{-1}$ for TSS, as well as hydraulic retention times of at least two days in fully vegetated zone (ITRC, 2003; USEPA, 2000). In practice, hydraulic retention times of five days have been used to treat food processing wastewater to reach secondary standards even with higher TSS loadings (Bojcevska & Tonderski, 2007; Sohsalam et al., 2008; Yirong & Puetpaiboon, 2004). Heavily loaded (BOD >112 kg $ha^{-1}d^{-1}$) systems do not support growth of nitrifying organisms (ITRC, 2003). On a monthly average basis, with sufficient pretreatment and wetland area, tertiary discharge standards of less than 10 mg/L BOD, TSS, and total nitrogen can be met. Due to the internal load of leaf litter and detritus, wetlands that receive no wastewater loading will still discharge low levels of BOD, generally in the range of 2 to 12 mg/L (USEPA, 2000).

Fully vegetated FWS wetlands will not support nitrification unless sufficient open water areas for aeration are present, or the system is very lightly loaded. Furthermore, one to two growing seasons may be necessary to develop sufficient vegetation to support microbial nitrification. Reported removal efficiencies for total nitrogen have ranged from 33 to 45% (Seabloom & Hanson, 2005; Wass, 2006). However, with food processing wastewater that has been pretreated in aerobic lagoons or equivalent systems that offers pre-nitrification, total nitrogen removals of 72–92% (Burgoon et al., 1999; Sohsalam et al., 2008) and TKN removals of 49% (Yirong & Puetbaiboon, 2004) have been achieved in FWS wetlands. Wetland plants and organic carbon promote formation of anaerobic conditions and nitrogen removal through denitrification and harvesting of plants (USEPA, 2004). The growth of macrophytes may be inhibited by high concentrations (about 1,200 mg/L) of ammonia nitrogen (de Feo et al., 2005). Potential chemical amendments for the enhancement of denitrification include molasses and sodium tripolyphosphate (Roudebush & Beilke, 2006).

Dissolved phosphorus may constitute a large fraction of total phosphorus in food processing wastewater and it is easily released from FWS

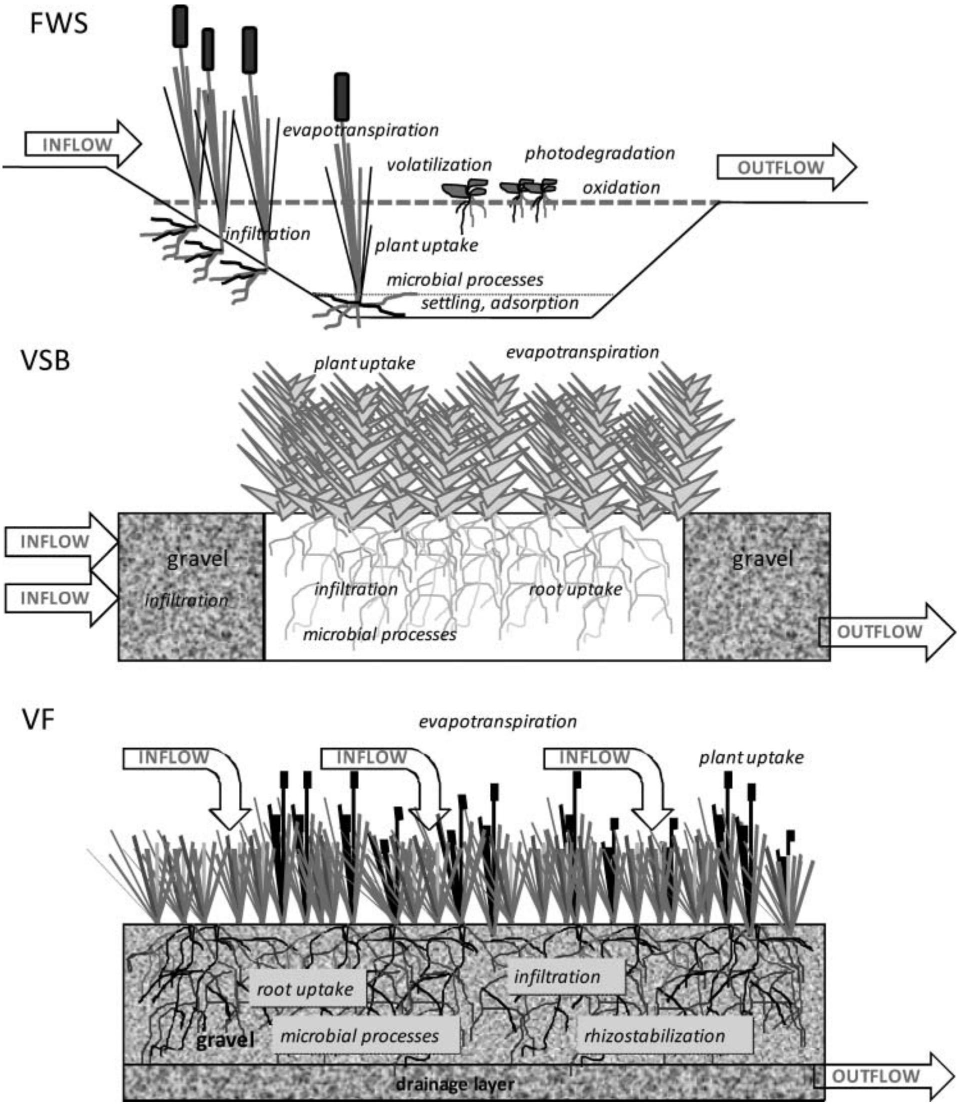


**FIGURE 1.** Generic illustrations of major types of constructed wetlands: free water surface (FWS), vegetated submerged bed (VSB), and vertical flow (VF) wetland, with pollutant removal processes indicated.

wetland (Bojcevska & Tonderski, 2007). Permanent phosphorus removal in FWS wetlands is small and is the result of adsorption to solids and plant detritus (ITRC, 2003). In an FWS wetland, about 20% of phosphorus stored in the wetland was buried in sediments, supporting a phosphorus removal rate of approximately 0.11 kg $ha^{-1}d^{-1}$ (40 kg $ha^{-1}$ $year^{-1}$, assuming constant rate) (Kadlec, 1997). Harvesting of vegetation could improve removal, considering that plant uptake rate could be about the same as determined for terrestrial plants, 84 kg $ha^{-1}$ $year^{-1}$ (Bojcevska & Tonderski, 2007; Crites et al., 2006).

## Vegetated Submerged Bed (VSB) Wetlands

Vegetated submerged bed (VSB) wetlands or horizontal-flow subsurface wetlands are also known as reed beds, rock-reed filters, gravel beds, and the root method. They are designed to keep the water level below the surface of a porous bed material which is planted with wetland vegetation (ITRC, 2003; Wallace, 2005).

TSS removal in VSB wetlands is good at loading rates up to 200 kg $ha^{-1}d^{-1}$ based on maximum monthly influent TSS. BOD removal is not as good as TSS removal, and it usually controls the design requirements to meet secondary standards of 30 mg/L. BOD loading rate should be less than 60 kg $ha^{-1}d^{-1}$ (USEPA, 2000). However, enhanced removal of BOD to 89% in food processing wastewater from an undefined source has been achieved with two subsequent wetland beds (Vrhoušek et al., 1996), and COD removal from winery wastewater has been nearly complete when using a recirculation system (Grismer et al., 2003).

VSB wetlands are well-suited for microbial removal of nitrate via denitrification. However, a source of biodegradable organic carbon may be a limiting factor, particularly for secondary-treated wastewater (Seabloom & Hanson, 2005; USEPA, 2000). Organic nitrogen from primary effluent is easily removed, as it is associated with suspended solids, and the remaining organic nitrogen is converted to ammonia by ammonification under anaerobic conditions. Ammonia removal through nitrification is inconsistent due to oxygen-limited conditions. If ammonia (or total Kjeldahl nitrogen, TKN) removal is required, another treatment process should be used in conjunction with a VSB system (USEPA, 2000). Nitrogen uptake by plants has been found to account for 11% of the nitrogen removal in a constructed wetland, and a maximum TKN storage of 55 mg/g dry weight has been measured in the leaves of the *Phragmites australis* seedlings (Browning & Greenway, 2003).

Phosphorous is partially removed in VSB wetlands, but the effectiveness decreases over time. In the long term, phosphorous removal by plant harvesting is limited to only about 0.5 kg $ha^{-1}d^{-1}$ (183 kg $ha^{-1}$ $year^{-1}$, assuming constant rate) or 3% of the total phosphorous removal, so VSB wetlands should not be expected to meet discharge standards for phosphorous (Browning & Greenway, 2003; USEPA, 2000). Furthermore, it has been shown that the removal of both phosphorus and nitrogen can be strongly impaired with increasing application rate (Lyon, 2006).

In the long term, decrease in treatment efficiency of food processing wastewater in VSB wetlands due to the formation of preferential flow pathways has been reported, especially in cold climates. However, supplemental aeration and the thermal protection given by vegetation help to maintain consistent treatment efficiency within the VSB (Grismer et al., 2003, Muñoz et al., 2006).

### Vertical Flow (VF) Wetlands

In vertical flow (VF) wetlands, wastewater is uniformly applied to the top of the wetland bed, and effluent is withdrawn via perforated pipes on the bottom. VF systems typically consist of two parallel sets of individual VF cells. When wastewater is applied in these sets in rotation, aerobic conditions can be restored during the drying period. Polishing of the effluent occurs in one or more horizontal-flow cells. The main advantage of the concept is the restoration of aerobic conditions during the periodic resting and drying period. This allows removal of BOD and ammonia nitrogen at higher rates than with FWS and VSB wetlands (Crites et al., 2006).

The VF system has a good efficiency and relatively small land requirement, and it is in use at several locations in Europe (Crites et al., 2006; de Feo et al., 2005). VF wetlands have been more successful than the other wetland types for ammonia reduction from landfill leachate (Crites & Plude, 2006). However, there is insufficient performance data for a rational (mathematical) design model that would allow calculation of theoretical BOD removal rate, required retention time, and other design parameters (Crites et al., 2006; de Feo et al., 2005).

As a complete purification sequence for raw dairy-cheese wastewater, Reeb and Werckmann (2005) have shown promising results with two subsequent VF wetlands followed by a VSB wetland (i.e., 90% BOD removal, 74% TKN removal, and 89% suspended solids removal).



### Advanced Wetland Systems

Rustige and Platzer (2001) have documented treatment of sewage with a multi-stage constructed wetland concept. The concept includes primary settling, vertical (VF) and horizontal flow (VSB) reed bed, followed by UV-disinfection and a special exchangeable phosphorus filter bed, filled with iron-enriched sand to maintain long-term capacity. In addition, denitrification can be improved by recycling the effluent back to the system. The median removal of total nitrogen from sewage within the soil filters can be estimated to be 37% at VF and 18% at VSB.

Reciprocating subsurface flow-constructed wetlands consist of paired wetland cells that are filled and drained on a frequent and recurrent basis. This operating technique turns the entire wetland system into a fixed-film biological reactor, in which it is possible to control redox potential in alternating aerobic and anaerobic zones. Variations of reciprocating systems have been under development since 1993, and their treatment potential has been demonstrated for food processing and many other waste streams. They can be retrofitted to existing conventional treatment systems and integrated with other wastewater treatment systems (Behrends et al., 2001).

For wastewater with high ammonia loading, ammonia removal can be improved by constructing a specific nitrification filter bed. However, there are some requirements. For example, BOD level must be low (BOD/TKN $<1$), surface must be moist and properly aerated all the time, and there should be at about 10 g of alkalinity/g ammonia (Crites et al., 2006).

## Environmental Impacts and Sustainability

Because of the many functions that a wetland system may have, it is particularly difficult to carry out a comprehensive life cycle analysis. Irrespective of the different ecological functions, the potential to reuse treated wastewater and nutrients accumulated in vegetation and sludge can be critical for assessing the economical and environmental benefits of wetland systems (Brix et al., 1999).



A number of environmental functions and benefits have been attributed to wastewater treatment at constructed wetlands (Day et al., 2004; ITRC, 2003; Sustainable Conservation, 2005; USEPA, 2004). FWS systems in particular have high ecological and recreational values. Wetlands can facilitate wastewater recycling and reuse by providing additional polishing to reclaimed water and serving as water reservoirs for irrigation. Recycled wastewater can be used to restore natural wetland ecosystems and create habitat for wetland-dependent plants and wildlife, particularly migratory birds. However, VSB wetlands provide little habitat as compared to FWS wetlands. Constructed wetlands help to control floods, surface runoff, and soil erosion, and protect downstream receiving waters. Aquaculture and agroforestry have been occasionally used to provide further reuse option for low-salinity wastewater. However, aquaculture involves several concerns about bioaccumulation of hazardous compounds and energy consumption (Brix et al., 1999). In any treatment wetland, contaminant accumulation must be monitored to maintain ecological health of the system (ITRC, 2003).

The advantages of VSB systems include increased treatment efficiencies in a small area, owing to the high surface area for bacterial biofilm growth. As compared with FWS wetlands, there are fewer pest problems, reduced risk of exposing humans or wildlife to toxics and pathogens, decreased waterfowl use, and less suitable habitat for mosquitoes and other vectors (ITRC, 2003; Wallace, 2005). Stakeholder concerns that treatment wetlands will attract endangered species can also be relieved (O'Brien et al., 2002).

Chemical usage is limited to occasional prevention of mosquitoes and other vectors, and waste production is low if the sludge can be used for soil improvement. When properly designed and maintained, treatment of wastewater in constructed wetlands does not threaten natural ecosystems. However, if constructed wetlands impact natural waters, discharges need to be regulated (ITRC, 2003). Native soil, imported clay, or geosynthetic liner may be required to isolate wetland from groundwater (Sustainable Conservation, 2005). Effluent quality from wetlands may be threatened by wildlife excreta and short-circuiting, which leads to shorter detention times (USEPA, 2004; WHO, 2006).

Constructed wetlands are generally viewed favorably by the general public and regulatory agencies (O'Brien et al., 2002; ITRC, 2003). If optimum conditions are maintained, nitrogen and BOD assimilation in wetlands will occur indefinitely, as they are primarily controlled by microbial processes and generate gaseous end products. In contrast, phosphorus assimilation is related to the adsorption capacity of the soil and long-term storage within the system that is often controlled by the redox chemistry (USEPA, 2004).

## POND SYSTEMS



### Stabilization Ponds

Pond treatment technology has been used for wastewater treatment from municipal, agricultural, and industrial sources, including sugar cane, distillery, seafood processing, palm oil, and starch industry, and several commercial applications are available (Calero et al., 2000; Onyia et al., 2001; Rajbhandari & Annachhatre, 2004; Shilton & Walmsley, 2005; Sirianuntapiboon & Srikul, 2006; Travieso et al., 2006). Advantages of pond systems include long retention times, which give them some capacity of buffering fluctuations in wastewater flow and load, particularly in summer when the treatment efficiency is at its highest (Calero et al., 2006). The costs of construction and maintenance of a stabilization pond system are lower than those of constructed wetlands, and minimal or no mechanical equipment is needed (Shilton & Walmsley, 2005). On the other hand, the capability of designing pond treatment for different types of wastewater is still impaired by the uncertainty about the processes occurring in stabilization ponds, including ammonia transformations, the effect of benthic feedback on oxygen demand and odor formation, and accumulation and degradation of sludge (Walmsley & Shilton, 2005).

Wastewater treatment in stabilization ponds is largely due to the growth of algae, which utilize nutrients, aerate the pond, and capture carbon dioxide, metals, and trace organics. In anoxic ponds, photosynthetic bacteria are essential degraders (Calero et al, 2000). Sunlight provides effective disinfection of pathogens and decomposition of organic compounds. In colder climates, ponds achieve at least preliminary settling (Shilton & Walmsley, 2005).

Different types of stabilization ponds vary in complexity, degree of inflow and discharge control, mixing, aeration, and composition of algal population. Oxidation ponds (or aerated ponds) and facultative ponds are the most common pond types. A reasonably well-established "standard pond system" for municipal wastewater treatment comprises several ponds in series, either (1) a primary facultative pond and a series of maturation ponds, or (2) an anaerobic pond, secondary facultative pond, and a series of maturation ponds (Shilton & Walmsley, 2005). Wastewater reservoirs can also be considered as treatment ponds. When operated in a batch mode, BOD and nutrient removal efficiencies similar to those in other types of treatment ponds have been reported in several studies (Juanicó, 2005).

Depending on the literature source, typical BOD loadings of 22–67 kg $ha^{-1}d^{-1}$ (Crites et al., 2006) or up to 100–400 kg $ha^{-1}d^{-1}$ (Mara, 2005) have been suggested for facultative ponds. In fact, ponds appear to perform better than constructed wetlands at BOD loading higher than 80 kg $ha^{-1}d^{-1}$, while at lower loadings they are equally efficient (Kadlec, 2005). Facultative ponds normally achieve 75–85% BOD removal and unfiltered effluent concentration of 50–70 mg/L for municipal wastewater. Due to the removal of biomass, BOD concentration in filtrated samples can be <20 mg/L and often <10 mg/L (Walmsley & Shilton, 2005). Further BOD removal can be achieved in maturation ponds, about 10–25% per pond. BOD removal in stabilization ponds is highly temperature-dependent (Mara, 2005). Nutrient removal in stabilization ponds is inconsistent. Based on several studies, total nitrogen removal efficiency in facultative ponds has been 20–80%, ammonium removal 20–95%, and total phosphorus removal 20–50%. Nutrient removal data from maturation ponds are scarce (Craggs, 2005b; Crites et al., 2006)

Disadvantages of stabilization ponds include high land area requirements. Particularly for facultative ponds, the size of the aerated surface is critical, whereas anoxic ponds require less space (Calero et al., 2000; Shilton & Walmsley, 2005). Ponds should be located in non-flooding, flat sites with low permeability, unless bottom liners are used (Crites et al., 2006). Basic maintenance operations include removal of sludge and floating scum, and insect control (Lloyd, 2005). Monitoring of cyanobacteria might also be appropriate due to their toxic metabolites (Oudra et al., 2000). Treated wastewater can be used for irrigation or discharged to surface water systems. Drip irrigation with pond effluent typically follows additional treatment to remove solids, and disinfection is needed prior to spray irrigation (Crites et al., 2006). Anaerobic ponds can be used for biogas generation (Shilton & Walmsley, 2005). Algal growth also provides a sink for the carbon dioxide produced by bacterial respiration (Rockne & Brezonik, 2006; Tadesse et al., 2004). Environmental impacts of stabilization ponds are similar to those of FWS wetlands.

There are several possibilities to improve treatment efficiency in pond systems. For example, floating elements can be used to improve hydraulic characteristics and algal attachment. Solids recycling reduces the need for sludge disposal and improves ammonium removal (Craggs, 2005b; Crites et al., 2006). Bubble aeration ensures year-round aerobic conditions throughout the pond and prevents settling of suspended solids (Liu, 2007). Many treatment processes comprise of a sequence of different types of ponds. For example, a sequencing batch reactor type pond system with aerobic and anaerobic lagoons has been used to the treatment of raw meatworks wastewater, yielding a 98% BOD removal and up to 95% N removal (Raper & Green, 2001). Advanced Integrated Wastewater Pond Systems (AIWPS®) are designed to achieve both efficient wastewater treatment and recovery of resources from the wastewater through capture of biogas, harvest of algal biomass as a fertilizer or feed, and reuse of treated effluent (Craggs, 2005a). The greatest advantage of AIWPS is considered to be the reduction in the need for sludge disposal as the influent organic carbon is assimilated by microalgae, and the algal biomass is harvested by dissolved air flotation or sedimentation. In practice, AIWPS have been applied for almost 30 years without disposing of the sludge. Furthermore, deep sludge blanket adsorbs potentially toxic compounds (Green et al., 1995).



## Evaporation Ponds

Evaporation ponds can be used for wastewater disposal and precipitation or volatilization of wastewater impurities but not specifically for treatment. Evaporation ponds are widely used for the disposal of saline agricultural drainage, spent brines from olive processing, and reject brines from desalination. They can compete very successfully with mechanical evaporation systems such as single-effect evaporators or vapor compression evaporators in zero-liquid discharge plants (Ahmed et al., 2000). About 3,400 hectares of evaporation ponds or basins are in operation in San Joaquin Valley and Tulare Lake Basin in Californian Central Valley, and this number may be increasing due to integrated farm drainage management systems (Bureau of Reclamation, 2006; Romero Barranco et al., 2001). Evaporation ponds are the least costly means of brine disposal in regions where the land costs are low, the climate sustains high evaporation rates, and other disposal (e.g., ocean) would entail higher transportation or piping costs (Ahmed et al., 2000). However, problems arise with high salt concentrations and higher viscosity (Romero Barranco et al., 2001).

Site selection for evaporation ponds is critical, especially if no bottom lining is used. Criteria that have been used for site selection include soil type, groundwater quality, land use, presence of endangered or protected species and habitats, flood risk, seismic risk, and proximity of suitable reuse areas for treated wastewater (Bureau of Reclamation, 2006).

Evaporation rate determines the surface area of the pond, whereas the depth should be adapted to the water storage requirements (Ahmed et al., 2000). Evaporation can be improved by various systems that increase the wind velocity, turbulence, exposed surface area, or vapor pressure difference between the surface and atmosphere (Ahmed et al., 2000; Romero Barranco et al., 2001). It has also been suggested that the absorption of solar radiation could be enhanced by adding a specific additive to the water or to the basin (García Marí et al., 2007). As salinity in the pond increases, evaporation rate decreases (Ahmed et al., 2000).

The major environmental impacts of evaporation ponds are associated with the open water surface (which attracts wildlife), potential of high-salinity wastewater to contaminate aquifers or surface water bodies, and formation of odorous compounds. In California, a solar evaporator should be designed to prevent standing water, mitigate for wildlife impacts, prevent migration of salt constituents into the vadose zone, and avoid nuisance from wind-blown salt spray or other source (Cismowski et al., 2006; State Water Resources Control Board, 2005). Unlined evaporation ponds have led to salinity increases in the underlying groundwater in Central Valley in California and many other locations. Currently, there are more stringent construction and monitoring requirements for groundwater protection and monitoring in California, and evaporation ponds are located in areas where underlying groundwater is not potable and not considered to be a source of drinking water (i.e., TDS >3,000 mg/L) (Cismowski et al., 2006; Romero Barranco et al., 2001; State Water Resources Control Board, 2005). For the protection of surface and groundwater, it is also important that the evaporation pond has sufficient storage volume and liners that are mechanically strong and impermeable (Ahmed et al., 2000). Formation of malodorous compounds, such as volatile fatty acids, during long-term storage of food processing wastewater can be avoided by adding nitrates when supplementary electron acceptors are needed for mineralization (Bories et al., 2005).

Closure of an evaporation pond after use may also pose an environmental and economical problem. From a regulatory point of view, there are three options available: (1) harvest of salts and other deposits followed by clean closure; (2) closure in place; (3) removal of salts/deposits and disposal in an authorized waste facility (State Water Resources Control Boards, 2005). Harvested salts can be disposed of to the sea or an approved waste disposal site, or sold for further reuse (Ahmed et al., 2000). There are several options available for salt recovery and reuse (Arakel et al., 2006). However, it has been difficult to find viable markets for harvested salts from food processing brines or to identify other sustainable final disposal option (Cismowski et al., 2006). Burial of precipitated salts in place should entail safety measures such as placement of low-permeability soil cap and long-term monitoring (Bureau of Reclamation, 2006).

## Salinity-Gradient Solar Ponds

A salinity-gradient solar pond is a collection and storage system for solar energy in which transfer of heat into the atmosphere is prevented by dense salt solution on the bottom of the pond. Salinity-gradient solar ponds can be used for generating heat and electricity, water desalination, and as a thermal energy storage that smoothens daily and seasonal cycles of solar irradiation. The energy of the sunlight that reaches the bottom of the pond remains entrapped there, because the high salinity in the bottom layer makes the water too heavy to rise to the surface, even when it is hot (Green Trust, 2007). There are naturally formed salinity-gradient lakes (such as the Aral Sea and soda lakes in various parts of the world), but salinity gradient can also be created by dissolving sodium chloride or other salt to the bottom layer, or by utilizing waste brines (Deambi, 2001; Hassairi et al, 2001). Salt is not consumed, so it can be reused in the same process or recovered from the bottom of the pond for other use, if fresh brines are continuously discharged to the pond. A large pond may contain up to 400 t of salts (Deambi, 2001).

The El Paso salinity gradient solar pond, in operation since 1985, has successfully demonstrated applications including desalination, waste brine management, industrial process heat production, and electricity generation, though currently at a limited scale. The experience of the El Paso plant and other projects has led to several advancements in optimizing the technology and improving heat recovery systems (Karakilcik et al., 2006; Lu et al., 2004).



## Solar Stills

Solar stills are distillation plants that utilize solar irradiation for the generation of clean water from brines. Hence, reuse value of wastewater is high, but so are the requirements for influent quality. Solar stills can be built in many different configurations. For example, a greenhouse-type flat solar still is simple to build and operate. It consists of a black lining and an airtight space in which evaporation and condensation occur simultaneously (United Nations, 2001). Solar distillation plants are not commercialized yet, except for a few individual units (Mathioulakis et al., 2007). However, solar stills might be suited particularly for remote arid or semi-arid regions and islands, where the application of complex desalination technologies may be economically or technically infeasible (United Nations, 2001). Furthermore, solar stills can be an ideal source of water in dry climates. Their use would be especially useful providing fresh water to crops growing under controlled-environment greenhouses and requiring little water. Besides, seasonal changes in solar still production match with changing water requirements of most crops (García Marí et al., 2007).

Experimental results and theoretical calculations show that fresh water could be produced at rate more than 40 L $m^2day^{-1}$ (Orfi et al., 2007). At the highest sunlight intensity studied by Voropoulos et al. (2001), 20.5 $MJ/m^2$, fresh water production rate was 18–25 L $m^2day^{-1}$ but it was increased to up to 40–53 L $m^2day^{-1}$ with solar collectors. In addition to solar collectors, efficiency of solar stills could be improved by using hot wastewater or systems such as multiple effect solar stills, cascade solar stills and inclined-type models (United Nations, 2001). Multiple effect solar stills can also be considered to represent direct processes utilizing humidification–dehumidification techniques through a broad area of design solutions (Mathioulakis et al., 2007).

As an example from the food processing industry, a distillate produced from olive mill wastewater was free from solids, and the chemical oxygen demand (COD) and TKN were 80% and 90% lower, respectively, than before the treatment. However, treatment capacity in the proposed system was low, about 3 L $m^2 day^{-1}$ (Potoglou et al., 2003).

## CONCLUSIONS



Land application, constructed wetlands, and pond systems comprise a variety of well-established but also more innovative designs for the exploitation of intrinsic biological, physical, and chemical treatment processes. Some of these natural treatment systems, particularly OF, FWS, and VSB, can be integrated with other natural systems, which makes it possible to establish complete treatment and disposal processes for selected food processing wastewaters and reclaim valuable natural resources (see Table 3). However, algal growth in stabilization ponds may impair their integration with other ponds and land application systems, unless a filtration step is used. Salinity-gradient solar ponds and in particular solar stills have fairly high requirements for the influent quality. Thus, they are suited for wastewater from which BOD, TSS, and nutrients have been removed, or for brines that have been segregated in food processing plant. Other treatment systems may require pre-filtration to reduce the load of solids.

Clearly, wastewater utilities have a lot of experience from the utilization of natural treatment and disposal systems, though most of it has not been consistently reported. The experience used to guide the design of natural treatment systems has mostly been gained from polishing of municipal wastewater. The problem with generalized design parameters is that in practice, the treatment systems must always be designed with respect to the actual wastewater volumes and concentrations, and predicted site-specific treatment capacity. The rate of most natural processes (possibly excluding adsorption/absorption and settling) is highly temperature-dependent. Furthermore, experience from real field sites shows that the expected quality has not always been achieved.

The quality of wastewater effluent from natural treatment systems is unlikely to meet the high standards set for water in contact with food or food-contact surfaces. Water use and reuse standards for activities associated

**TABLE 3.** Summary of the characteristics of natural wastewater treatment and disposal systems

| | Treatment systems | | | | | | Disposal systems* | | |
|---|---|---|---|---|---|---|---|---|---|
| | SR | SAT | OF | FWS | VSB | Stabilization pond (SP) | Evaporation pond (EP) | Salinity-gradient pond | Solar still |
| Pretreatment for† | None | (EP) | SR, SAT, FWS, VSB, EP | SR, SAT, OF, (SP), EP | SR, SAT, OF, (SP), EP | FWS, VSB, (OF) | | | |
| BOD removal | High | High | Low/moderate | High | High | Moderate/high | | | |
| TSS removal | High | High | Moderate/high | High | High | Low | | | |
| N removal (main process) | High (nitrification, denitrification) | Moderate (denitrification) | Moderate/high (nitrification) | Low (nitrification) | Variable (denitrification) | Variable | | | |
| P removal | High (limited)‡ | High (limited)‡ | Low | Low/moderate | Low/moderate | Variable | | | |
| TDS removal | None | None | None | None | None | None | High | High | High |
| Water reuse | High§ | Moderate | Moderate | High | High | Moderate | None | Moderate | High |
| Nutrient reuse | High | Moderate | Low | Low | Low | Moderate | Low | | |
| Specific product or beneficial use | Crops | Groundwater recharge | Surface water recharge | Recreational area | — | Biogas | (Salts) | Energy, water, (salts) | Water,(salts) |
| Lifetime | Long | Moderate | Moderate | Moderate | Moderate | Long | Long | Long | Moderate |
| Operational period | Warm | Warm, wet/dry cycles | Warm | (Any) | (Any) | (Any) | Warm, dry | Warm, dry, sunny | Warm, sunny |

*Classification as disposal systems based on the lack of treatment capacity for BOD, TSS, and nutrients. Can also be considered as polishing (tertiary treatment).
†Potentially suitable for pretreatment of a given technology.
‡High efficiency, but total capacity is limited.
§For influent, but no reuse for effluent.

with the production of meat and poultry products, irrigation of food crops, and direct or indirect potable reuse can also be restrictive (California Code of Regulations, Title 22; K-State Research and Extension, 2002; WHO, 2006). However, there are other, less critical options for the beneficial reuse of food processing wastewater. If the benefits of wastewater treatment and reuse are viewed from regional perspective, it can be more sustainable to reuse food processing wastewater in the vicinity of the production facility than aim at zero-discharge processes within the facility. When properly maintained, natural treatment systems produce water that is generally considered to be suitable for restricted urban or recreational use, irrigation of non-food crops or food crops that are commercially processed, construction, environmental (e.g., wetlands), and selected industrial uses. For these purposes, BOD and TSS concentrations should be smaller than 30 mg/L (USEPA, 2000).

Compared to municipal wastewater, food processing wastewater may contain even higher BOD and TSS concentrations but smaller concentrations of pathogens, heavy metals, and various pharmaceuticals and personal care products that have been found to accumulate in sewage. Thus, reuse of food processing effluents may provide a safer alternative to the reuse of municipal wastewater in water-deficient regions. However, pathogens and metals can be a problem in some food processing wastewaters and for selected beneficial uses. The absence or sufficient removal of pathogens, antibiotics, and hormones should be guaranteed prior to the treatment of wastewater of animal origin. Components of food processing wastewater that are generally not regulated but may impair beneficial uses include alkalinity/hardness, sulfates, sugars, oils and fats, pesticides, detergents, arsenic, iron, and manganese (State Water Resources Control Board, 2005). The long-term response of natural systems to these components and the need to segregate certain wastewater streams should be carefully evaluated.

Natural treatment systems trade land for energy: land requirement is high but energy consumption low. They also have various environmental impacts, both positive and negative, as summarized in Table 4. The overall feasibility of natural systems for a food processor is highly dependent on local conditions, such as hydraulic and climatic conditions and the cost and availability of suitable treatment sites. Ideally, natural treatment systems tolerate relatively high variations in wastewater flux and composition, and they give added value through resource recovery and reuse. From an ecological point of view, natural treatment systems can be viewed as a tool to restore natural wetland ecosystems and maintain productivity of agricultural areas depleted of organic matter and nutrients. Natural assimilation capacity can even be temporarily exceeded, provided that there is a sufficient recovery period (e.g., minimal wastewater discharge in winter) and groundwater quality is not endangered.

**TABLE 4.** Summary of potential environmental and aesthetic impacts

| Land application | Wetlands | Ponds |
|---|---|---|
| Negative impacts | | |
| • Wetting and clogging of soil | • Pollutant accumulation in biota | • Large area requirements |
| • Changes in vegetation type | • Rodents, mosquitoes, birds | • Flooding risks |
| • Salinity impacts on vegetation | • Spreading of water hyacinth, etc. | • Sludge accumulation |
| • Surface water eutrophication | • Contamination with wildlife excreta | • Contamination with wildlife excreta |
| • Groundwater contamination | • Limited phosphorus binding capacity | • Pollutant accumulation in wildlife |
| • Leaching of organic matter and metals* | • Limited salinity removal | • Limited phosphorus binding capacity |
| • Odors* | • Groundwater contamination | • Groundwater contamination |
| • May cause erosion | | • Odors |
| | | • Toxic algal blooms |
| Positive impacts | | |
| • Increase in microbial activity | • Wastewater reuse by irrigation | • Utilization of solar energy |
| • Increase in fertility and productivity | • Groundwater replenishment | • Utilization of biogas |
| • Recycling of phosphorus | • Recreational values | • Water reservoir |
| • Immobilization of metals | • Ecosystem and habitat restoration | • Salinity removal |
| • May prevent erosion | • Flood and erosion control | • Wastewater volume reduction |
| • Less wastewater transportation | • Surface runoff control (Aquaculture, agroforestry) | • Algal consumption of $CO_2$ |

*With high organic loading (anaerobic conditions).

## ABBREVIATIONS

| | |
|---|---|
| AIWPS | advanced integrated wastewater pond system |
| BOD, $BOD_5$ | biological oxygen demand |
| COD | chemical oxygen demand |
| FWS | free water surface (surface-flow wetland) |
| IFDM | integrated on-farm drainage management |
| OF | overland flow (land application method) |
| SAR | sodium adsorption ratio |
| SAT | soil-aquifer treatment (land application method) |
| SR | slow rate (land application method) |
| TDS | total dissolved solids |
| TKN | total Kjeldahl nitrogen |
| TSS | total suspended solids |
| VF | vertical flow (wetland) |
| VSB | vegetated submerged bed (horizontal-flow subsurface wetland) |



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