LITERATURE REVIEW – Optimization Methods and Best Management Practices for Facultative Lagoons

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Table of Contents

1.0 INTRODUCTION 1
1.1 Regulatory Background 2
2.0 WASTEWATER TREATMENT PROCESSES 4
2.1 Carbon 5
2.2 Nitrogen and the Nitrogen Cycle 5
2.3 Phosphorus 8
2.4 Total Suspended Solids 10
2.5 References 10
3.0 LAGOON INVENTORY 11
4.0 LITERATURE REVIEW 12
4.1 Operational Improvements 12
4.1.1 Maintenance BMP's/Activities 12
4.1.1.1 Retention Times/Fill-Draw 13
4.1.1.2 Solid Waste Removal 16
4.1.1.3 Reduce Influent 21
4.1.1.4 Mosquitos/Insects 27
4.1.1.5 Maintenance Schedules/Activities 37
4.1.2 Lagoon Covers 41
4.1.3 Floating Barriers 45
4.2 Mechanical Techniques 54
4.2.1 Aeration 54
4.2.2 Fixed-film Bioreactors 60
4.2.3 Phosphorous Removal 79
4.2.4 Alternate Carbon Sources 85
4.2.5 Pathogens/Disinfection 92
4.3 Ecological Approaches 96
4.3.1 Constructed Wetlands 96
4.3.2 Vermiculture 112
4.3.3 Vegetation Harvesting 117
4.3.4 Floating Islands 130

4.3.5 Terrestrial Application 140
4.3.6 Other Ecological Technologies 149
4.3.6.1 Fish 149
4.3.6.2 Eco-Machines™ 150
4.4 Technology Summary Table 153
5.0 RECOMMENDED TECHNOLOGIES/BMPS 153

Appendix

Appendix A – Numeric Nutrient Criteria for Montana (DEQ-12A) and EPA Approval
Appendix B – Summary of Facultative Lagoons in Montana
Appendix C – Facultative Lagoon Technology Summary Table

List of abbreviations:

Al- aluminum

BER – Board of Environmental Review

BOD - Biochemical oxygen demand

BMPs – Best Management Practices

°C – Celsius

CIPP - Cured-In-Place Pipe

CCTV - Closed Circuit Television

Ca - calcium

$CaCO_3$ - calcium carbonate

CEP - Controlled Eutrophication Process

CH_4 - methane

COD - chemical oxygen demand

CO_2 - carbon dioxide

DAF - dissolved air floatation

DBP - disinfection byproducts

DO – dissolved oxygen

EBPR - enhanced biological phosphorus removal

EDI - Environmental Dynamics International

°F – Fahrenheit

FC - fecal coliform

Fe - iron

FeOOH - iron oxide

FLUMP - Floating Lagoon Pumper

gpcd - gallons per capita per day

gpd – gallons per day

H_2S – hydrogen sulfide

HDPE – High Density Polyethylene

HRT- hydraulic residence time

I&I – infiltration and inflow

IDEAL - Intermittently Decanted Extended Aeration Lagoon

IFAS - Integrated Fixed-lm Activated Sludge

LPR - Lemna Polishing Reactor

MBRR - Moving Bed Bio-Reactors

MDEQ – Montana Department of Environmental Quality

Mg - magnesium

MGD – million gallons per day

Mg/kg – milligrams per kilogram

Mg/l – milligrams per liter

MnO_2 - manganese dioxide

MPDES - Montana Pollutant Permit Discharge Elimination System

N_2 - nitrogen gas

NH_3 – ammonia

NH_4^+ - ammonium

NO_2^- - nitrite

NO_3^- - nitrate

NNC - numeric nutrient criteria

NTF - nitrifying trickling filter

O_2- oxygen

O&M - operations and maintenance

ORP - oxidation-reduction potential

OTE - oxygen transfer efficiency

P – Phosphorus

PAOs - polyphosphate accumulating organisms

PAR - photosynthetically active radiation

PAS - Partitioned Aquaculture System

PHA - poly-L-hydroxyalkanoates

polyP - polyphosphate

PO_4^{3-} - phosphates

PTO - power-take-off

PVC - Polyvinyl chloride

RBC - Rotating Biological Contactors

redox - reduction-oxidation

ROL - Root Oxygen Loss

rpm - rotations per minute

SAGR® - Submerged Attached Growth Reactor

SARs - Submerged Aerated Reactors

$SO_4$2- - sulfates

TAN – Total Ammonia Nitrogen

TKN - -total Kjeldahl nitrogen

TOC - total organic carbon

TN – Total Nitrogen

TP – Total Phosphorus

TSS - Total suspended solids

UV - ultraviolet

µg/l - micrograms per liter

VFAs - volatile fatty acids

WWTP – Wastewater treatment plant

1.0 INTRODUCTION

In response to new numeric nutrient standards adopted by the Board of Environmental Review (BER), The Montana Department of Environmental Quality (MDEQ) commissioned this report to identify available technologies, Best Management Practices (BMPs), and optimization methods for increasing ammonia (NH_3), total nitrogen (TN), and total phosphorus (TP) removal efficiencies for facultative lagoon systems in Montana.

Facultative lagoons are the most common configuration for small community wastewater applications. These lagoons have an operating depth of 4 to 8 feet deep and have detention times greater than 30 days. "Facultative" means that both aerobic and anaerobic conditions are present. A facultative lagoon system forms 3 layers with respect to dissolved oxygen (O_2). The top layer is aerobic, the bottom layer is anaerobic, and the middle is facultative. Facultative bacteria are responsible for most of the treatment that occurs in these ponds. In facultative lagoons, suspended solids settle to the bottom of the pond where there is less dissolved oxygen (DO). The facultative bacteria will become anaerobic to digest these solids. Circulation in the pond will eventually bring these bacteria to the surface where high levels of DO exist. When this occurs, the bacteria will become aerobic to stabilize the dissolved organics that are present.

Odors from these types of lagoons are normally not objectionable. As the hydrogen sulfide (H_2S), generated in the anaerobic zone, rises to the surface the DO present oxidizes it into sulfates (SO_4^{2-}) that do not cause odors. The algae that grow in the lagoon are critical to the successful stabilization of the organic load. They supply 70 to 80% of the oxygen that the bacteria will need. The rest of the oxygen comes from wind and wave action on the surface. The aerobic decomposition in the upper zone results in the uptake of DO and the release of carbon dioxide (CO_2). The algae will take in carbon dioxide and through photosynthesis use it to create sugars and release DO that is used by the aerobic bacteria.

The report includes the following major topics:

- Regulatory overview of newly adopted nutrient standards;
- Background information on nutrient processes that drive the wastewater treatment process;
- An inventory of facultative lagoons in Montana;
- Comprehensive literature review of technologies and BMPs for facultative lagoon optimization; and
- Technology evaluation and recommendation of the most promising technologies and BMPs for Montana lagoons.

The goals of this project are to provide thorough review of the technical literature regarding optimization methods and BMPs available for reducing ammonia, total nitrogen, and total phosphorus concentrations, and identify the most promising technologies for future application in Montana. A variety of literature was consulted including the primary scientific and engineering literature as well as the gray literature which was likely to contain studies, technical reports, and company brochures describing the application of lagoon optimization methods and BMPs. Emerging, innovative technologies were also reviewed. All technologies were evaluated in their overall ability to remove ammonia, total nitrogen, and total phosphorus, as well as site specific limitations and performance criteria related to Montana.

1.1 Regulatory Background

Surface water nutrients standards in Montana have been recently updated, specifically, ammonia, total nitrogen, and total phosphorus. New ammonia criteria have also been released by the Environmental Protection Agency (EPA 2013), which are more stringent than the current Montana standards (MDEQ 2012). Montana has yet to adopt the EPA's updated ammonia criteria. The revised nutrient criteria from EPA are shown in Table 1 below.

Table 1. Comparison of past and current EPA-recommended aquatic life water quality criteria magnitudes for Total Ammonia Nitrogen (TAN) measured in milligrams per liter (mg/L) at pH 7 and 20°C.

Criterion Duration	1999 Criteria	2009 Draft Updated Criteria	2013 Final Updated Criteria
Acute (1-hour average)	24	19	17
Chronic (30-day rolling average)	4.5*	0.91*	1.9*
* Not to exceed 2.5 times the criterion continuous concentration as a 4-day average within a 30-day period			
Criteria frequency: Not to be exceeded more than once in three years on average.			

NOTE: A link to the 2013 EPA Final Ammonia Criteria is located at: http://water.epa.gov/scitech/swguidance/standards/criteria/aqlife/ammonia/index.cfm.

In addition to updating its national ammonia criteria recommendations, EPA has recently approved new numeric nutrient criteria (NNC) for total nitrogen and total phosphorus as submitted by MDEQ and adopted by the Montana BER (EPA 2015). The new NNC range in concentration from 25 to 150 micrograms per liter (µg/L) total phosphorus and 275 to 1300 µg/L total nitrogen. The standards were developed for all wadeable streams based on Omerinik Ecoregions (Level III and IV) and apply from July 1 through September 30 each year (see Endnote 1). The NNC for the Yellowstone River were developed using a mechanistic model (QUAL2K) and apply from August 1 through October 31. Reach specific nitrogen and phosphorus criteria for several segments in the Gallatin watershed were developed to reflect known geologic conditions in the area and apply from July 1 through September 30. Table 2, which is taken directly from Circular DEQ 12-A (Table 12A-1) summarizes Montana's NNC (MDEQ Circular 12-A).

Table 2. Base numeric nutrient standards for wadeable streams in different Montana ecoregions. If standards have been developed for level IV ecoregions (subcomponents of the level III ecoregions) they are shown in italics below the applicable level III ecoregion. Individual reaches are in the continuation of this table.

Ecoregion[1,2](level III or IV) and Number	Ecoregion Level	Period When Criteria Apply[3]	**Numeric Nutrient Standard[4]**	
			Total Phosphorus (µg/L)	Total Nitrogen (µg/L)
Northern Rockies (15)	III	July 1 to September 30	25	275
Canadian Rockies (41)	III	July 1 to September 30	25	325
Idaho Batholith (16)	III	July 1 to September 30	25	275
Middle Rockies (17)	III	July 1 to September 30	30	300
Absaroka-Gallatin Volcanic Mountains (17i)	IV	July 1 to September 30	105	250
Northwestern Glaciated Plains (42)	III	July 1 to September 30	110	1300
Sweetgrass Upland (42l), Milk River Pothole Upland (42n), Rocky Mountain Front Foothill Potholes (42q), and Foothill Grassland (42r)	IV	July 1 to September 30	80	560

Northwestern Great Plains (43) and Wyoming Basin (18)	III	July 1 to September 30	150	1300
River Breaks (43c)	IV	See Endnote 5	See Endnote 5	See Endnote 5
*Non-calcareous Foothill Grassland (43s), Shields-Smith Valleys (43t), Limy Foothill Grassland (43u), Pryor-Bighorn Foothills (43v), and Unglaciated Montana High Plains (43o)**	IV	July 1 to September 30	33	440

*For the Unglaciated High Plains ecoregion (43o), criteria only apply to the polygon located just south of Great Falls, MT.

[1] See Endnote 1 [2] See Endnote 2

[3] See Endnote 3 [4] See Endnote 4

Table 2, Continued. Base numeric nutrient standards for individual wadeable streams (and wadeable-stream reaches), and large-river reaches.

Individual Stream or Reach Description[2]	Period When Criteria Apply[3]	**Numeric Nutrient Standard[4]**	
		Total Phosphorus (µg/L)	Total Nitrogen (µg/L)
Wadeable Streams: Clark Fork Basin			
Flint Creek, from Georgetown Lake outlet to the ecoregion 17ak boundary (46.4002, -113.3055)	July 1 to September 30	72	500
Wadeable Streams: Gallatin River Basin			
Bozeman Creek, from headwaters to Forest Service Boundary (45.5833, -111.0184)	July 1 to September 30	105	250
Bozeman Creek, from Forest Service Boundary (45.5833, -111.0184) to mouth at East Gallatin River	July 1 to September 30	76	270
Hyalite Creek, from headwaters to Forest Service Boundary (45.5833,-111.0835)	July 1 to September 30	105	250
Hyalite Creek, from Forest Service Boundary (45.5833,-111.0835) to mouth at East Gallatin River	July 1 to September 30	90	260
East Gallatin River between Bozeman Creek and Bridger Creek confluences	July 1 to September 30	50	290
East Gallatin River between Bridger Creek and Hyalite Creek confluences	July 1 to September 30	40	300
East Gallatin River between Hyalite Creek and Smith Creek confluences	July 1 to September 30	60	290
East Gallatin River from Smith Creek confluence mouth (Gallatin River)	July 1 to September 30	40	300
Large Rivers[6]:			
Yellowstone River (Bighorn River confluence to Powder River confluence)	August 1 – October 31	55	655
Yellowstone River (Powder River confluence to stateline)	August 1 – October 31	95	815

[2] See Endnote 2 [6] See Endnote 6

[3] See Endnote 3

[4] See Endnote 4

(1) Ecoregions are based on the 2009 version (version 2) of the U.S. Environmental Protection Agency maps. These can be found at: http://www.epa.gov/wed/pages/ecoregions/mt_eco.htm. For Geographic Information System (GIS) use within the Department, the GIS layers may be found at: L:\DEQ\Layers\Ecoregions.lyr

(2) Within and among the geographic regions or watersheds listed, base numeric nutrient standards of the downstream reaches or other downstream waterbodies must continue to be maintained. Where possible, modeling methods will be utilized to determine the limitations required which provide for the attainment and maintenance of water quality standards of downstream waterbodies.

(3) For the purposes of ambient surface water monitoring and assessment only, a ten-day window (plus/minus) on the beginning and ending dates of the period when the criteria apply is allowed in order to accommodate year-specific conditions (an early-ending spring runoff, for example).

(4) The 30 day average concentration of these parameters may not be exceeded more than once in any five-year period, on average.

(5) In this level IV ecoregion, the narrative standard for nuisance aquatic life (ARM 17.30.637(1) (e)) applies in lieu of specific base numeric nutrient standards. 7

(6) Table E-1 below shows the beginning and ending locations for large rivers in Montana.

River Name	Segment Description
Big Horn River	Yellowtail Dam to mouth
Clark Fork River	Bitterroot River to state-line
Flathead River	Origin to mouth
Kootenai River	Libby Dam to state line
Madison River	Ennis Lake to mouth
Missouri River	Origin to state line
South Fork Flathead	Hungry Horse Dam to mouth
Yellowstone River	State-line to state-line

(7) (7) No lake or reservoir in Table12A-2 shall have a total nutrient concentration that exceeds the values shown, as an annual average, more than once in any three year period, on average. The Department will determine on a case-by-case basis whether or not a permitted discharge to a stream or river is likely to be affecting any downstream lake or reservoir. If so, the permittee would be required to meet its average monthly nutrient limit year-round.

(8) Parameters listed under this column are standards specific to lakes and reservoirs.

(9) Standards and related assessment information (excluding Secchi depth) are to be determined from 0- 30 m depth-integrated samples. Samples and Secchi depth measurements are to be collected at the Midlake Deep site which is located approximately 1 mile west of Yellow Bay Point in a pelagic area of the lake (approximately at latitude 47.861, longitude -114.067).

The new total nitrogen and total phosphorus standards include provisions for public dischargers to apply for a variance for total nitrogen, total phosphorus, or both in order to develop nutrient reduction steps to comply with the new standards over time. Per §75-5-313 [8], Montana Code Annotated (MCA), general nutrient standards variance may be established for no more than 20 years. The intent of establishing nutrient reduction steps upfront for most of the 20 year period is to provide permittees regulatory certainty well out into the future. This in turn allows for better facility planning and financing (MDEQ 2014).

MDEQ Department Circular DEQ-12A, and the EPA approval letter and associated rationale for its approval, is included as Appendix A. Guidance documents related to the updated NNC are Nutrient Variances (MDEQ Circular DEQ-12B, July 2014), and Base Numeric Nutrient Standards Implementation Guidance (MDEQ 2014, Version 1.0) which are located at: http://deq.mt.gov/wqinfo/Standards/default.mcpx.

2.0 WASTEWATER TREATMENT PROCESSES

This section discusses the primary water quality constituents and associated chemical processes that are the basis of all wastewater treatment technologies discussed in this literature review; as a result, they have been included as a separate section. The type and level of wastewater treatment is closely related to the standards and/or expectations set for the effluent quality. Wastewater treatment is categorized as primary, secondary, and tertiary. Primary treatment removes larger particles by screening, grit removal, and sedimentation. Secondary treatment involves biological processes that remove organic matter and smaller suspended solids. Tertiary treatment again uses biological processes to remove nutrients and fine particles (Sen et al. 2013). Many of the technologies in this literature review

incorporate two or more treatment levels. The various treatment processes within those levels reduce the following constituents that effect receiving water quality:

- Carbon, or biodegradable organics, which can serve as "food" for microorganisms in the receiving water body. Microorganisms combine this matter with oxygen from the water to yield the energy they need to thrive and multiply; unfortunately, this oxygen is also needed by fish and other aquatic organisms in the river. Heavy organic pollution can lead to "dead zones" where fish and aquatic organisms cannot survive; sudden releases of heavy organic loads can lead to fish kills.
- Nutrients, including nitrates (NO_3^-) and phosphates (PO_4^{3-}) can lead to high concentrations of unwanted algae, which can themselves become heavy sources of biodegradable organic load. Treatment processes may also neutralize or remove industrial wastes and other toxic chemicals.
- Pathogenic bacteria and other disease causing organisms, which are most relevant where the receiving water is used for drinking or recreation, or where people would otherwise be in close contact with it.
- Total suspended solids (TSS) are physical particles that can clog rivers or channels as they settle under gravity.

2.1 Carbon

Biological oxygen demand (BOD_5) and carbonaceous biological oxygen demand ($CBOD_5$) are methods of measuring the amount of organic carbon in an aquatic system. Removal of BOD from the water column can occur via settling of sediments and through microbial activity in both aerobic and anaerobic environments (Dodkins and Mendzil 2014). Aerobic heterotrophs use organic carbon to produce cell tissue (IWA 2000) and require approximately 1.5 kilograms (kg) of oxygen for every kg of BOD to break down organic carbon in a wastewater system (EPA 2002). Nitrifying bacteria also require a carbon source, but are outcompeted by heterotrophs and thus not considered responsible for reduction of BOD. For the highest rate of BOD reduction, heterotrophic organisms must be present. Heterotrophs thrive in underwater bacterial films attached to rocks, plants, plant parts, soils, and other surfaces. Anaerobic heterotrophs can also break down organic carbon in anaerobic conditions but at a slower rate than aerobic conditions (IWA 2000). Thus, there are various mechanisms to remove organic carbon from a wastewater system with the most efficient and effective being the use of heterotrophs in an aerobic aquatic environment.

2.2 Nitrogen and the Nitrogen Cycle

Nitrogen (N) is present in many forms in a domestic wastewater treatment plant. Removing nitrogen from wastewater primarily occurs through microbial processes. Microbial respiration occurs via different methods depending on the reduction-oxidation (redox) potential of the environment. Redox potential describes the likelihood of a transfer of an electron from a compound of a higher energy state (often organic carbon) to a compound with a lower energy state. This transfer is induced by microbes. The microbes use the energy produced from this transfer for cell growth and reproduction (Faulwetter 2009). Environments with a high redox potential have an abundance of compounds with low energy states, such as oxygen, and promote aerobic processes such as nitrification where ammonium (NH_4^+) is converted to nitrate

(NO_3^-). Aerobic environments are defined as those with free oxygen available. Environments with a low redox potential do not have an abundance of low energy state compounds and promote anoxic process such as denitrification where nitrate is converted to nitrogen gas (N_2). In anoxic environments free oxygen is not available; however, bound oxygen is available. Environments with extremely low redox potential have an even lower abundance of low energy state compounds and promote anaerobic processes such as methanogenesis where carbon dioxide is converted to methane (CH_4). In anaerobic conditions neither free oxygen nor bound oxygen is available (Dodkins and Mendzil 2014). In both anoxic and aerobic processes other compounds are substituted for oxygen as an energy source. The oxygen compound releases the most energy when reduced, followed by nitrate, manganese dioxide (MnO_2), iron oxide (FeOOH), sulfate, and finally carbon dioxide. Because microbes use the energy produced in these reaction for cell growth and reproduction, reactions that produce more energy are more desirable for microbial populations. High redox waters are often associated with oxygenated waters, but can also be characterized by other compounds that can oxidize (Dodkins and Mendzil 2014).

Step 1: Ammonification. Ammonification is the conversion of organic nitrogen to ammonium. Most nitrogen excreted by humans is in the form of organic nitrogen (dead cell material, proteins, and amino acids) and urea. In the absence of oxygen (as it travels through sewer pipes), this organic nitrogen is converted to ammonium by microorganisms. As a result, the majority of the influent nitrogen is ammonia, although some organic nitrogen remains. In most treatment plants, less than 2 mg/L of organic nitrogen passes through the treatment plant untreated. The rest is converted to ammonia. Ammonification is mostly an anaerobic process, sometimes called hydrolysis. Most treatment processes do nothing to enhance organic nitrogen removal; it is not managed. However, treatment facilities with total nitrogen effluent limits can often times reduce the organic nitrogen to less than 1 mg/L by subjecting wastewater to strongly anaerobic and organically-rich conditions.

Step 2: Nitrification. Nitrification is the conversion of ammonium to nitrate. In the presence of oxygen (aerobic conditions), bacteria will break ammonium down to nitrite (NO_2^-) and then to nitrate. Nitrifying bacteria are slower growing and more sensitive to environmental upset than BOD removing bacteria. Generally, nitrification occurs only under aerobic conditions at DO levels greater than 1.0 mg/L.

The nitrification process produces acid. The acid lowers the pH of the aqueous environment and is, unless buffered, toxic to the nitrifying bacteria. If there isn't enough alkalinity present in the wastewater, bacteria will not complete the nitrification process; nearly all of the ammonia will be converted to nitrite but not all of the nitrite will be converted to nitrate.

Nitrate can have serious human health effects if consumed in drinking water. Nitrate, other forms of nitrogen and phosphorus can have deleterious effects on the environment, as excess nitrogen stimulates the process known as eutrophication in surface water bodies. For this reason, many alternative technologies have been designed to remove total nitrogen from wastewater. These technologies use bacteria to convert ammonia and nitrate to gaseous nitrogen. In this form, nitrogen is inert and may be released to the air.

There are several key factors that must be met in order for the nitrification process to occur. The following is an excerpt from whitepaper titled *Biologically Removing Ammonia-Nitrogen in Cold Water Lagoon Systems* (Triplepoint Environmental 2014):

> *1. Oxygen Supply*
> *Nitrification consumes large quantities of oxygen. For every pound of BOD oxidized, 1.5 lb/ O_2 are consumed. For every pound of ammonia oxidized, 4.6 lb/ O_2 are consumed (Metcalf & Eddy). In addition, in order for uninhibited nitrification to occur, a minimum working dissolved oxygen (DO) level of 2.0 mg/L is required and a DO level of 5 mg/L is optimal.*
>
> *2. BOD Loading*
> *Nitrifying bacteria do not compete well against BOD consuming heterotrophic bacteria, which survive on the waste in the water. For nitrification to take place, the BOD must be sufficiently reduced to eliminate competition. Generally a BOD level of 20–30 mg/L is ideal (OHM 2009).*
>
> *3. pH & Alkalinity*
> *"Nitrification is pH-sensitive and rates decline significantly at pH values below 6.8. At pH values near 5.8 to 6.0, the rates may be 10 to 20 percent of the rate at pH 7.0" (U.S. EPA, 1992). "Optimal nitrification rates occur at pH values in the 7.5 to 8.0 range."*
>
> *4. Water Temperature*
> *Temperature is the hardest factor to control, and therefore the most problematic parameter for achieving nitrification in wastewater lagoons located in colder climates. Following the van't Hoff-Arrhenius equation for temperature dependence of the reaction rate and empirical data, the most optimal range is 82°F to 97°F. An acceptable range can be maintained as low as 68°F, or 20°C. As temperatures decrease, nitrifying bacteria slow down, and nitrification efficiency drops significantly until halting completely at 32°F. "Knowles, Downing & Barrett (1965) have estimated that the temperature coefficient (increase in maximum specific growth rate constant) for Nitrosomonas was 9.5% per degree centigrade. The temperature coefficient for Nitrobacter was found to be about 5.9% per degree centigrade" (Sharma, 901).*
>
> *5. Nitrifier Mass*
> *There must be a sufficient population of nitrifying bacteria present in order to nitrify. This nitrifying mass is referred to as "attached growth organisms," which means that these organisms thrive in environments where they can attach themselves to the surface of an object or substrate. In lagoons, such objects include the lagoon slopes, baffles and even perhaps algae particles.*
>
> *6. Mixing*
> *Ammonia can be released from sludge accumulation anaerobically digesting at the bottom of a lagoon. In extreme cases where there is insufficient mixing and sludge levels are not properly managed, ammonia effluent levels can be higher than that of influent. Ideally, it is recommended that sludge depths remain below two feet (OHM 2009).*

Another adverse effect of a poorly mixed lagoon is "Short Circuiting". This occurs when a basin becomes stratified, allowing influent flows to take a "short cut" through it by only moving through the top layer (or stratum) of the water. This lack of homogeneity results in reduced retention time for the water and generally leads to poor overall treatment (OHM 2009).

7. Toxins
Nitrifying bacteria are more easily inhibited by compounds such as heavy metals than are the BOD-reducing bacteria. See Sharma, page 909 for a complete list of inhibitors. High levels of toxins in wastewater lagoons can prevent nitrifying bacteria from multiplying in sufficient quantity to accomplish the desired levels of nitrification.

Step 3: Denitrification. Denitrification is the conversion of nitrate to nitrogen gas. The biological reduction of nitrate to nitrogen gas is performed by bacteria that live in a low-oxygen environment. To thrive, the bacteria need a soluble carbon source (BOD). Particulate BOD needs to be broken down into solution before it can be used by the denitrifiers. Denitrifying organisms are generally less sensitive to toxic chemicals than nitrifiers, and recover from toxic shock loads quicker than nitrifiers. However, most wastewater treatment facilities have more difficulty with denitrification than ammonia removal for two principal reasons: 1) At low temperatures, it becomes more difficult to drive down the DO concentration and keep the oxidation-reduction potential (ORP) values at desired negative millivolt levels; and 2) Variations in BOD loadings also make it difficult to maintain consistent nitrate removal. Denitrifying bacteria require a considerable amount of soluble BOD and many facilities find it difficult to provide an ongoing supply of readily digestible BOD.

2.3 Phosphorus

Phosphorus removal from the aquatic environment can occur through plant uptake, microbial and algal uptake, sorption, adsorption, or precipitation. Permanent, long term total phosphorus removal from a water column is difficult because all of these processes are finite, or will re-release phosphorus under certain conditions. Sediments, plants, microbes, and algae all act as sediment sinks for phosphorus (P), but this phosphorus will eventually be re-released when these organisms die unless harvested (Dodkins and Menzil 2014).

Phosphorus plant uptake varies by plant species but is minimal (Garcia et al. 2010). Phosphorus plant uptake is controlled by climate and season to some degree in that dormant plants are not actively metabolizing nutrients. Phosphorus will re-enter the aquatic system when plants die and degrade on site. Plant harvesting is not typically effective in cooler climates because as plants senesce, nutrients are transferred to the roots, which remain even after harvest (Garcia et al. 2010; White and Cousins 2013).

Nutrient uptake by microbes, algae and periphyton is similar to plant uptake in that nutrients are used for energy and cell growth, but after die-off, the phosphorus returns to the system (Garcia et al. 2010). In low phosphorus systems periphyton and algal communities can be an effective means of phosphorus uptake. When the biomass dies, it ends up on the sludge surface, or as particulate in effluent (Reddy et al. 2000). In larger, more mechanical and

controlled environments, a process called enhanced biological phosphorus removal (EBPR) is used where microbes called polyphosphate accumulating organisms (PAOs), are encouraged to uptake phosphorus in a more efficient manner. There are two separate processes within EBPR and the microbiological details are not well understood. The first step takes place in an anaerobic environment where PAOs use organic substances to produce poly-L-hydroxyalkanoates (PHA) which can be stored as an energy source. To produce PHA the PAOs use polyphosphate (polyP) as an energy source and this actually releases phosphate into the water column. The second step takes place in an aerobic environment after all the carbon has been removed from the system. Organisms then use the stored PHA as an energy source in place of carbon to synthesize polyP which uses phosphate in greater quantities than it was released in the first step. Under these contrived conditions the PAOs accumulate phosphorus in excess of normal scenarios in order to continue metabolic growth. In a more mechanical wastewater treatment plant these POAs are then collected with the sludge, removing phosphorus from the water column (Seviour et al. 2003). According to Garcia and others (2010), only one initial pilot scale test has been performed to see if EBPR can be performed in a less controlled setting such as a wetland (Garcia et al. 2010). The study showed promise, but little other research further supports this idea.

Phosphorus adsorption is the physical adherence of phosphorus to plant material or other organic matter (Garcia et al. 2010). Phosphorus can also be adsorbed into certain soils. In acidic soils phosphorus will bond with oxides and hydroxides of iron (Fe) and aluminum (Al) if available. In alkaline soils phosphorus will bond with calcium carbonate ($CaCO_3$) if available (Reddy et al. 2000). Wastewater is generally more alkaline and thus calcium carbonate content will likely be more influential than iron and aluminum (Arias et al. 2001). The use of specific media that is fine grained and high in iron, aluminum and/or calcium (Ca) can increase the rate and volume of phosphorus sorption (IWA 2000; Garcia et al. 2010). In addition to particle size and soil content, conditions such as pH, redox potential and temperature also influence the rate of sorption (Reddy et al. 2000; Arias et al, 2001). Once phosphorus occupies all available surface area, phosphorus sorption will cease. The length of time it takes for phosphorus to occupy all available surface area can be either weeks or years depending on the material (Garcia et al. 2010). Reddy and others (2000) reviewed two studies testing various medias for phosphorus removal. One study found fly ash to have the highest phosphorus-adsorption in contrast to bauxite, shale, burnt oil shale, limestone, zeolite, and light expanded clay. Another study comparing sands found Wollastonite to be effective because of its calcium content. Still another study found that steel slag removed phosphorus for up to 5 years at a retention ratio of 1230 milligram/kilogram (mg/kg) (Garcia et al. 2010). Organic matter and oyster shells have also been shown to increase phosphorus sorption (Garcia et al. 2010). Jenssen et al. (2005) suggest naturally iron or calcium rich sand as well as manufactured aggregates can provide 15 years of greater than 90% removal of phosphorus if provided in adequate quantities (approximately 7 to 9 cubic meters/person). Prior to occupying all available surface area, phosphorus removal can cease and be desorbed in periods of high flows actually contributing phosphorus back to the water column (Garcia et al. 2010).

Precipitation of phosphorus from the water column is often lumped with sorption because of the difficulty of distinguishing one process from the other in the laboratory and field. However, precipitation is a separate process that occurs in aerobic conditions where phosphorus forms an insoluble compound by binding with certain metals. Precipitation appears to be the most effective long-term solution for phosphorus removal as most adsorption studies document adsorption for only up to one year (Garcia et al. 2010). In acidic conditions phosphorus will bind with iron and aluminum; in alkaline conditions it will bind with calcium and magnesium (Mg) (Reddy et al. 2000). Still, if exposed to anaerobic conditions, these compounds will dissolve and re-release phosphorus in the form of ortho-phosphate ions (Reddy et al. 2000). Dodkins and Mendzil (2014) suggest dredging lagoon sludge or wetland substrate every 10 years to sustain phosphorus removal. Garcia and others (2010) and Jenssen and others (2005) suggest using the removed material as a fertilizer if within health safety limits. Much of the accumulated phosphorus is plant available.

2.4 Total Suspended Solids

Total suspended solids (TSS) include all particles suspended in water which will not pass through a filter. TSS is present in sanitary wastewater and many types of industrial wastewater. There are also non-point sources of TSS, such as soil erosion from agricultural and construction sites.

As levels of TSS increase, a water body begins to lose its ability to support a diversity of aquatic life. Suspended solids absorb heat from sunlight, which increases water temperature and subsequently decreases levels of DO (warmer water holds less oxygen than cooler water). Some cold water species, such as trout and stoneflies, are especially sensitive to changes in DO. Photosynthesis also decreases, since less light penetrates the water. As less oxygen is produced by plants and algae, there is a further drop in DO levels.

TSS can also destroy fish habitat because suspended solids settle to the bottom and can eventually blanket the river bed. Suspended solids can smother the eggs of fish and aquatic insects, and can suffocate newly-hatched insect larvae. TSS can also harm fish directly by clogging gills, reducing growth rates, and lowering resistance to disease. Changes to the aquatic environment may result in diminished food sources, and increased difficulties in finding food. Natural movements and migrations of aquatic populations may be disrupted. For point sources, adequate treatment is necessary to insure that suspended solids are not present at levels of concern in waters of the state. Treatment typically consists of settling prior to discharge of the wastewater. Settling allows solids to sink to the bottom, where they can be removed.

2.5 References

Arias, C.A., M. Del Bubba, and H. Brix. 2001. Phosphorus removal by sands for use as media in subsurface flow constructed reed beds. Water Res. 35:1159–1168.

Dodkins, I and AF Mendzil. 2014. Enterprise Assist: Floating Treatment Wetlands (FTWs) in Water Treatment: Treatment efficiency and potential benefits of activated carbon. SEACAMS Swansea University. Prepared for FROG Environmental Ltd, Ban y Berlan, Llansadwrn, Llansadwrn, SA19 8 NA.

Faulwetter, Jennifer L., Vincent Gagnon, Carina Sundberg, Florent Chazarenc, Mark D. Burr, Jacques Brisson, Anne K. Camper and Otto R. Stein. 2009. Microbial processes influencing performance of treatment wetlands: A review. *Ecological Engineering*, Vol. 35, pp. 987-1004.

Garcia, Joan, Diederick P. L. Rousseau, Jordi Morato, Els Lesage, Victor Matamoros and Josep M. Bayona. 2010. Contaminant removal processes in subsurface-flow constructed wetlands: A review. *Critical Reviews in Environmental Science and Technology,* Vol. 40, pp. 561-661.

IWA. 2000. Constructed wetlands for pollution control: processes, performance, design and operation. IWA Publishing. London, UK. Available at: http://www.scribd.com/doc/208541250/Constructed-Wetlands-for-Pollution-Control-Processes-Performance-Design-and-Operation-by-IWA#scribd. Accessed March 11, 2015.

Jenssen, Peter D., Trond Maehlum, Tore Krogstad and Lasse Vrale. 2005. High performance constructed wetlands for cold climates. *Journal of Environmental Science and Health*, Vol. 40, pp. 1343-1353.

Knowles, Downing, and Barrett. 1965. Determination of kinetic constants for nitrifying bacteria in mixed culture, with the aid of an electronic computer. J Gen Microbiol. 1965 Feb;38:263-78.

Reddy, K.R., E.M. D'Angelo and W.G. Harris. 2000. Chapter 4:Biogeochemistry of Wetlands. In CRC Press. *Handbook of Soil Science*. Editor-in Chief, M.E. Sumner. Pp. G89-119.

Sen, Bulent, Mehmet Tahir Alp, Feray Sonmez, Mehmet Ali Turan Kocer, and Ozgur Canpolat. 2013. Relationship of algae to water pollution and waste water treatment. In *Water Treatment,* (ed. Walid Elshorbagy and Rezaul Kabir Chowdhury), ISBN 978-953-51-0928-0. InTech, DOI: 10.5772/51927. Available at: http://www.intechopen.com/books/water-treatment/relationship-of-algae-to-water-pollution-and-waste-water-treatment. Accessed May 7, 2015.

Seviour, Robert J., Takashi Mino and Motoharu Onuki. 2003. The microbiology of biological phosphorus removal in activated sludge systems. Federation of European Microbiological Societies (FEMS), Microbiology Reviews. Vol. 27, pp. 99-127.

Triplepoint Environmental. 2014.

White, Sarah A. and Mathew M. Cousins. 2013. Floating treatment wetland aided remediation of nitrogen and phosphorus from simulated storm water runoff. *Ecological Engineering*, Vol. 61, pp. 207-215.

3.0 LAGOON INVENTORY

According to data provided by MDEQ, Montana has a total of 108 facultative lagoons, with 3 of those being accelerated facultative lagoons. Accelerated facultative lagoons use an outside energy source to accelerate the facultative processes in the lagoon. The remaining 105 lagoons are traditional facultative lagoons. The bullet list below summarizes population characteristics of the facultative lagoons in Montana. A complete list of facultative systems is included in Appendix B.

- Largest population served by a facultative lagoon – 5,286 (Sidney).

- Largest population served by an accelerated facultative lagoon – 3,171 (Cut Bank).
- Smallest population served by a facultative lagoon – 65 (Pearson Addition, Cascade County and Trailer Terrace Mobile Home Park).
- Smallest population served by an accelerated facultative lagoon – 186 (Kevin).
- Total population served by both accelerated and facultative lagoons – 51,931.
- Average population served – 481.

4.0 LITERATURE REVIEW

This section includes a review of available technical literature related to optimization methods, technologies, and other BMPs for reducing ammonia, total nitrogen, and total phosphorus in facultative lagoon discharges. Literature types include: scientific and engineering reports and published texts, studies, white papers, and vendor information related to lagoon optimization and nutrient removal.

The literature review is organized into three primary categories: Operational, Mechanical, and Ecological. Each technology review includes a narrative description that addresses several key criteria, including: treatment parameters, design variations, operation and maintenance, constructability, relative costs, and others as appropriate and available. At the end of each review is a full list of references.

4.1 Operational Improvements

Operational activities include implementation of inspection, sampling, operation, or maintenance protocols that can lead to improved lagoon performance. This category includes an array of potential BMPs that do not require major design changes or capital expenditures. As a result these improvements are generally more cost-effective; however, they may have limited treatment capabilities and removal efficiencies.

Ideally operational improvements would be the first step in the lagoon optimization process. One-time or recurring maintenance or management improvements can have a significant impact on lagoon performance and may reduce or eliminate the need for significant capital improvements. *Wastewater Lagoon Troubleshooting: An Operators Guide to Solving Problems and Optimizing Wastewater Lagoon Systems* (H&S Environmental 2003) contains additional detailed information on operational improvements for lagoons.

4.1.1 Maintenance BMP's/Activities

There are several well documented maintenance activities and BMPs that should be part of a thorough routine maintenance plan for any facultative lagoon. These activities can include the following BMPs: sludge removal, vegetation control (aquatic and terrestrial), adjustment of flow patterns, burrowing animal control, infiltration/inflow, and others. Many of the major maintenance activities are included in this *document* as standalone optimization BMPs. Completing routine maintenance of the system will help maintain treatment effectiveness and avoid potentially large capital replacement or

maintenance in the future. Properly documenting all maintenance activities is critical for cost tracking, regulatory requirements, and proper planning.

4.1.1.1 Retention Times/Fill-Draw

General Description

A few documents are available pertaining to facultative lagoon retention time, fill, and draw. Generally all three of these parameters are addressed with the design of a facultative lagoon.

Issues Addressed

Retention Times: Longer detention times result in the decomposition of organic wastes and reduction of pathogens (EPA 2015).

Fill and Draw: Maintaining an effective pond depth is critical to overall system operation. Timing of fill and draw of ponds is critical to meet water quality objectives. Minimum and maximum depths must not be exceeded.

Design Variations

Retention Times:

Typical detention times are 20 to 180 days (EPA 2015) for loading rates 13 to 71 lbs/acre/day with longer times in colder climates. In these cases discharge may not be allowed during winter months (EPA 2015). According to Montana Circular DEQ-2, facultative lagoons must have a detention time of 40 to 80 days in the primary cells. Depending on the disposal method, the minimum detention range from 90 to 180 days.

The EPA recommends several technologies to divide lagoons into at least three cells for improved retention time and prevention of short circuiting of wastewater flows including: baffles, curtains, and multiple ponds in series or parallel (EPA 2015).

Adding baffles to cells reduces short circuiting and decreases the mean velocity per baffle section which will increase detention time. Alternating baffles should be 70% of the pond width and at least 2 baffles should be used at even spacing along the length of the pond with properly located inlets and outlets at opposite corners. This will increase the flow path length, retention time, and eliminate short circuiting (H&S Environmental 2003). Stub baffles that are 15% of the length of the pond and placed 20% of the length away from the inlet and outlet can improve flow for inlets and outlets on the same side of the pond (H&S Environmental 2003).

Rearranging inlet and outlet configuration including distributing inflows on the inlet end of the pond will reduce short circuiting and increase retention time. Locating inlets and outlets on opposite ends is a popular method to assist with distributing flow evenly and eliminating dead spots and short circuiting (H&S Environmental 2003).

Retention time can be affected by poor design: improper influent and transfer piping arrangement, dead zones, improper mixing from poor orientation with prevailing winds, sludge accumulation, stratification, too few ponds online, and infiltration and inflow (I&I) (H&S Environmental 2003).

Fill and Draw:
Pond levels should always be at least 2 to 3 feet deep. Excessive drawdown creates shallow areas that will allow for potential weed growth. Strategic drawdown during long freezing periods may kill certain undesirable aquatic vegetation (H&S Environmental 2003).

Lagoons should be sized to maintain a depth of 2 to 5 feet. The minimum depth of 2 feet is needed to prevent dry out of sludge, maintain aerobic conditions, and prevent rooted plant growth. The maximum depth of 5 feet is recommended to maintain the anaerobic, intermediate, and aerobic layers (Hygnstrom et al. 2010).

Wastewater treatment is generally poor in the summer and winter. Residence times are longer in the colder months when ponds are only filled. Lagoons are generally designed so that the critical detention time – winter – is met. Operational strategies vary and that could affect actual detention times. Usually wastewater treatment (carbon and nitrogen treatment) is better in the summer. During these times, discharge only occurs when water quality meets water quality objectives. Sampling of the wastewater can help determine whether treated wastewater can be discharged during the summer, residence times are lower as treatment is at the highest but algae blooms can make meeting discharge requirements difficult due to high TSS levels (WDNR 2013). High flows in receiving waters may make meeting discharge requirements easier as wastewater discharge will have less of an impact.

During spring and fall, the receiving waters are colder and have more DO. Wastewater can be drawn from the lagoon into the receiving water. The spring discharge can occur after the ice melts and BOD levels are acceptable. Fall discharges are generally the highest quality as summer treatment is at the highest level and algae blooms have subsided (WDNR 2013).

In multiple pond systems, the pond is isolated at least one month before the discharge period. The water quality is tested and if discharge restrictions are met, the water is discharged to the receiving water. The pond can be drawn to the minimum level of two feet (Circular DEQ-2).

A typical fill draw schedule for a two pond system (in series) in northern climates is as follows (WDNR 2013):

- May 1 – September 30: manage pond #1 to be no deeper than 6 feet, no discharge from pond #2
- October 1 – October 31: no discharge from pond #1, drawdown pond #2 to 2 feet

- November 1 – November 15: open structure from pond #1 to pond #2 to equalize pond #1 with pond #2, no discharge from pond #2
- November 15 – April 30: no discharge from pond #1 to pond #2, fill pond #2 and do not discharge

Operations and Maintenance Issues

Other than operations and maintenance (O&M) costs for operation of a lagoon (see Maintenance Schedule and Activities summary), this area does not have specific O&M issues.

Pond levels are usually controlled with a type of hydraulic structure; these can be simple flashboard weirs or valves as headworks to piping between ponds. Valves and other operational parts of structures should be inspected and exercised to ensure operable condition (WDNR 2013).

Relative Costs

Properly designing a pond system before initial pond construction (location of inlet and outlet structures, number of ponds, and baffles) is cheaper and more effective than trying to retrofit the system for optimal performance.

Constructability

Relocating inflow and outflow structures for maximum retention time and elimination of short circuiting is not cost effective when compared to adding baffles or other devices to improve treatment.

New/Innovative Approaches

Wastewater treatment improvements with regards to retention time, short circuit elimination, and fill draw schemes are unique to individual systems and can be easily addressed with proper design or retrofitting, adding additional components (floating barriers, covers, etc.), and proper maintenance of the system to improve treatment.

Recommendations

Improving system performance by optimizing retention time and fill/draw schemes is a well-known and low cost operational BMP for lagoons; however, design and implementation are unique to each individual system. Multiple optimization strategies were discussed in this section, which can typically be managed as part of regular lagoon operations. Combining these technologies with other technologies discussed in other sections will improve treatment.

References

U.S. Environmental Protection Agency (EPA). 2015. Facultative Lagoon Wastewater Fact Sheet.
http://water.epa.gov/scitech/wastetech/upload/2002_10_15_mtb_faclagon.pdf

Federation of Canadian Municipalities and National Research Council (FCMNRC). 2004. Optimization of Lagoon Operation.

https://www.fcm.ca/Documents/reports/Infraguide/Optimization_of_Lagoon_Operations_EN.pdf

H&S Environmental. 2003. Wastewater Lagoon Troubleshooting: An Operators Guide to Solving Problems and Optimizing Wastewater Lagoon Systems.

Hygnstrom, et al. 2010. Residential Onsite Wastewater Treatment: Lagoon Design and Construction, University of Nebraska-Lincoln Extension, Institute of Agriculture and Natural Resources, March 2010.
http://ianrpubs.unl.edu/live/g1441/build/g1441.pdf

Montana Department of Environmental Quality (MDEQ). Montana Circular DEQ-2

Wisconsin Department of Natural Resources (WDNR). 2013. Wastewater Operator Certification, Introduction to Ponds, Lagoons, and Natural Systems Study Guide.
http://dnr.wi.gov/regulations/opcert/documents/WWSGPondsINTRO.pdf

4.1.1.2 Solid Waste Removal

General Description

Although facultative wastewater treatment lagoons are intended to accumulate sludge to allow the biological decay of the settled material, too much sludge can negatively impact lagoon performance. The accumulation of solids in wastewater lagoons can affect effluent quality by reducing capacity and creating preferential flowpaths, thus reducing treatment efficiency. Lagoon sludge also releases nutrients that promote algal growth, and sludge may dislodge and enter the effluent; both adding to TSS and BOD in effluent streams.

Issues Addressed

Periodic de-sludging of a wastewater treatment lagoon is generally required to return it to its design capacity. Removing sludge increases performance by maintaining adequate retention time and reducing preferential flowpaths. This review evaluates several methods for de-sludging lagoons including: the use of traditional farm or construction equipment, agitation/pumping equipment, and cutter suction dredges.

Design Variations

There are four basic techniques for solids and sludge removal or reduction:

1. Agitation,
2. Drying,
3. Dredging, and
4. Enhanced biodegradation.

Agitation is the practice of re-suspending settled solids from the sludge blanket into the aerobic and facultative zones of a wastewater lagoon, increasing

biodegradation. An agitator/pump system is typically a power-take-off (PTO) system with hydraulic controls that can be operated as an attachment to a farm tractor. If used strictly for agitation, the system is equipped with recirculation nozzles that return the agitated solids to the lagoon.

Figure 1. Hydraulically driven agitator with recirculation nozzle.

Lagoons with sufficient capacity and multiple cells can be de-sludged by isolating a cell and draining it, allowing settled sludge to dry, and removing sludge using a front end loader or similar equipment. Leaving the sludge in the cell over the winter will allow it to freeze and concentrate for easier handling in the spring (Federation 2004).

The Kansas Department of Health and Environment (KDHE YEAR) provides guidance on desludging using this method. According to the guidance, dewatering should include isolating a lagoon cell and allowing its contents to "age" for one week, then gradually draining or pumping the "clear" water down over five to ten days, and finally pumping remaining liquid fraction to an active cell in the lagoon system. Once the sludge is exposed, it should be allowed to drain, then turned using farm or construction equipment to cultivate the dry crust and expose wet sludge underneath. Sludge should be turned at least twice per week until it is dry enough to remove and dispose.

If the lagoon configuration, time, or economic constraints preclude isolating and draining individual cells, or if traditional farm or construction equipment is unavailable or cannot access a lagoon without causing significant damage, sludge may be removed using equipment such as agitator/pump systems or floating dredges (Federation 2004).

Agitator/pump systems remove solids using the agitator equipment described above, except the agitated solid/liquid is pumped directly to dewatering equipment, drying beds, or tankers for land application.

Several dredging systems designed for use in wastewater treatment lagoons are commercially available. Similarly to agitator/pump systems, dredging systems

slurry the solid sludge with liquid contents and pump the material to tankers or dewatering equipment at the shoreline.

Self-propelled cutter suction dredges include: the Floating Lagoon Pumper (FLUMP), available for sale or rent by SRS Crisafulli of Glendive, Montana; the Piranha LS-Series Wastewater and Sewage Dredge manufactured by Piranha Pumps and Dredges of Albuquerque, New Mexico; and several turn-key dredging service providers around the United States. The SRS Crisafulli FLUMP is depicted below.

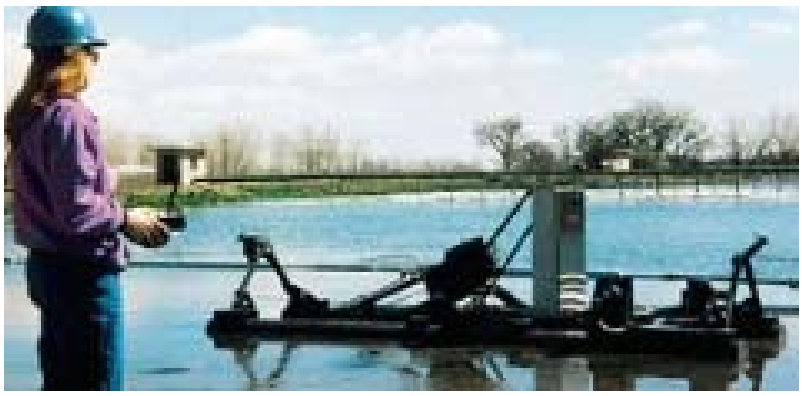

Figure 2. FLUMP with Radio Remote Control System

Enhanced biodegradation of lagoon solids, or biodredging, utilizes commercially available cultures of microorganisms and/or probiotics to lagoon sludges to degrade or promote the degradation of volatile solids. Biodredging and other alternative sludge removal techniques are discussed below in the New/Innovative Technologies section.

Operations and Maintenance Issues

When utilizing dredging or agitation/pumping systems, care must be taken not to scour the lagoon liner. The cutter heads and propellers can damage clay or synthetic liners, and recirculation nozzles can scour lagoon embankments. Floating debris should be removed from the lagoon to the extent practicable, as this material can damage the agitator.

Removal of partially dried sludge from a drained lagoon cell still produces sludges with relatively high water content. The literature describes facilities where cells that have been drained may take several weeks to dry sufficiently for subsequent excavation and removal, even with bi-weekly turning of the sludge to facilitate drying. Other facilities report that after the initial drying period, the excavated sludges are placed in separate drying beds for up to three years before disposal.

Relative Costs

Removal of saturated or nearly saturated sludge using excavation, agitation, or dredging equipment can result in large volumes of relatively heavy material and increased transportation and disposal costs, and passing the paint filter test is

typically required for landfilling of solid waste. Direct land application of slurried sludge can reduce these costs; however, regulatory constraints often preclude this option. Specialized, mobile dewatering equipment like centrifuges, belt presses, and filter presses can be employed to concentrate solids and potentially reduce transportation and disposal costs.

Costs to remove and dispose using mechanical techniques are estimated at approximately $100 per dry ton, resulting in total costs in the hundreds of thousands to over one million dollars for older municipal systems.

Constructability

The sludge removal techniques included in this review are generally easy to implement, with the exception of mechanical removal using front end loaders or track mounted equipment. This technique is not feasible in single cell lagoons that cannot be taken off line, or lagoons with appreciable sludge thicknesses that cannot be effectively dried for handling.

New/Innovative Approaches

Bio Energizer is a commercially available nutrient additive manufactured by Probiotic Solutions since 1989. Marketing materials state that the additive has reduced sludge thicknesses in aerated lagoons by up to 20 inches after 4.5 months (Pribiotic 2008), and a white paper reports the product is approximately 1/10th the cost of mechanical sludge removal (Probiotic 2013).

Water & Waste Digest 2014 reported that as much as a 31% reduction in sludge blanket thickness was achieved over a 4 month period by implementing Blue Frog Technology, which uses low energy circulators to create radial outflowing currents and floating baffles to redirect the horizontal flow downward to the anaerobic zone (Bettle 2014).

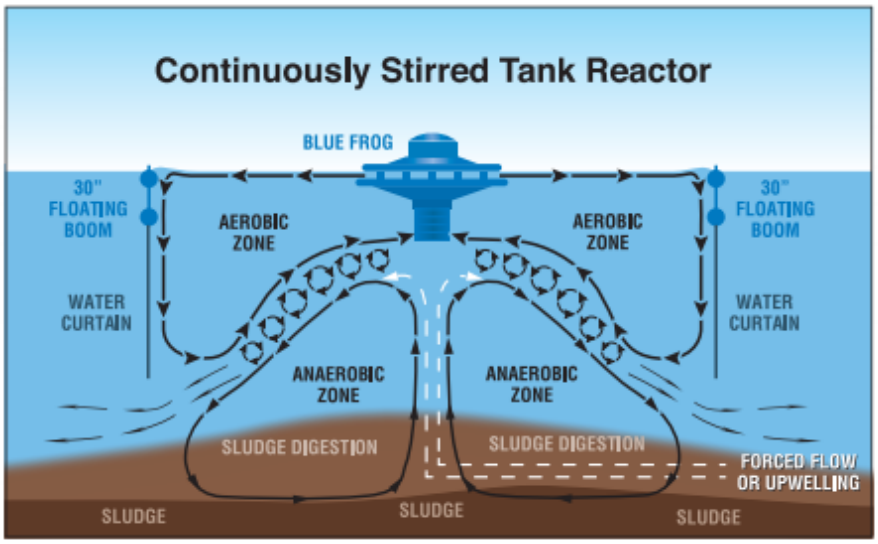

Figure 3. Blue Frog Technology (Bettle 2014)

Recommendations

If the objective for a treatment lagoon is to maintain the proper capacity for maximum treatment efficiency, periodic sludge removal is required. Mechanical removal techniques are proven technologies that are fully scalable to varying lagoon sizes, easy to implement, require no additional land or permanent infrastructure, and are 100% effective at removing solids. However; mechanical removal can cost in the hundreds of thousands of dollars and may be prohibitive to small municipalities.

Due to their apparently lower cost and ease of implementation, it is recommended that lagoon operators consider utilizing biodredging or systems like Blue Frog. The literature suggests that these systems not only remove existing sludge, but also slow the accumulation of solids for the life of the lagoon. Since it involves only the addition of a (typically) powdered bacterial culture, biodredging is easy to implement, requires no additional land or infrastructure, and is scalable to meet the needs of varying populations and lagoon sizes. It is, however, only moderately efficient at solids removal since only a finite amount of settled solids can be biodegraded, even in ideal conditions, and some fraction of inert material will always remain in the lagoon.

Although Blue Frog is a mechanical system, it works as a passive solids removal device that mixes lagoon contents and promotes biodegradation. While capital costs are reportedly much lower than those of one-time dig and haul, or dredging events, Blue Frog requires permanent infrastructure including the device itself and an electrical power source. Operation and maintenance costs are obviously higher as a result. Blue Frog is less easily scalable, as large lagoons may require multiple devices and, like biodredging, Blue Frog technology simply enhances biodegradation and therefore is not 100% efficient at removing solids. However, there is evidence where similar devices were installed to replace traditional aeration systems resulted in solids transfer issues from one cell to the next, substandard mixing, and inadequate oxygen supply.

References

Bettle. 2014. Biological Sludge Control, Eliminating Solids Accumulation in Wastewater Treatment Lagoons, Water & Wastes Digest, September 2014.

Federation of Canadian Municipalities and National Research Council (Federation). 2004. Optimization of Lagoon Operation, A Best Practice by the National Guide to Sustainable Municipal Infrastructure, August 2004.

KDHEa. Kansas Department of Health and Environment; Guidance for Desludging Domestic Wastewater Lagoons by Dredging Sludge from the Cell.

KDHEb. Kansas Department of Health and Environment; Guidance for Desludging Domestic Wastewater Lagoons by Drying the Sludge Before Removal.

Probiotic. 2013. Probiotic Solutions, It's Time to Rethink Biostimulants.

Probiotic. 2008. Bio Huma Netics, Inc. How to Cut Wastewater Treatment Plant Costs and Optimize Your System.

4.1.1.3 Reduce Influent

General Description

Reduction of non-sewage influent is a longstanding and common operational tool used by wastewater treatment operators that have problems with capacity exceedances, especially during wet weather conditions. Influent flow reduction can be identified through multiple lines of evidence and achieved through various technologies that are specific to the source of inflow.

Non-sewage influent into the wastewater collection system or treatment cells can be a significant problem for lagoon operators. Non-sewage influent enters wastewater systems as either ground water or storm water. The term to describe these inflows is infiltration and inflow (I&I). *Infiltration* is ground water that enters into the collection system through cracks or joints in the pipe. *Inflow* is extraneous water that enters the system through manhole covers, roof drains, lateral connections, foundation drains, or illicit connections. There may also be areas where the storm water and sanitary systems are combined. Additional influent flows generally lead to increased treatment volumes and costs, as once this non-sewage influent enters the system it becomes contaminated with wastewater and must also be treated.

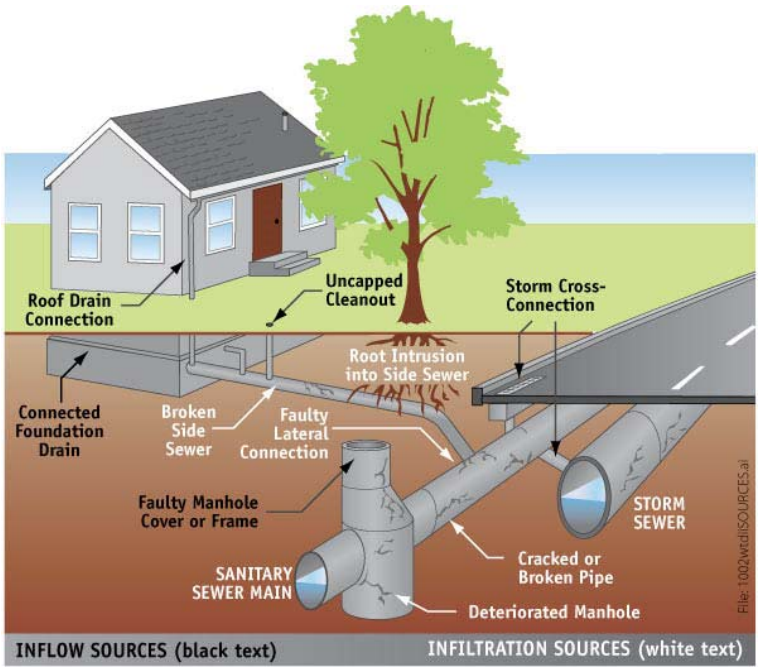

Figure 4. Infiltration and Inflow Sources. Photo courtesy of King County Department of Natural Resources and Parks (2014).

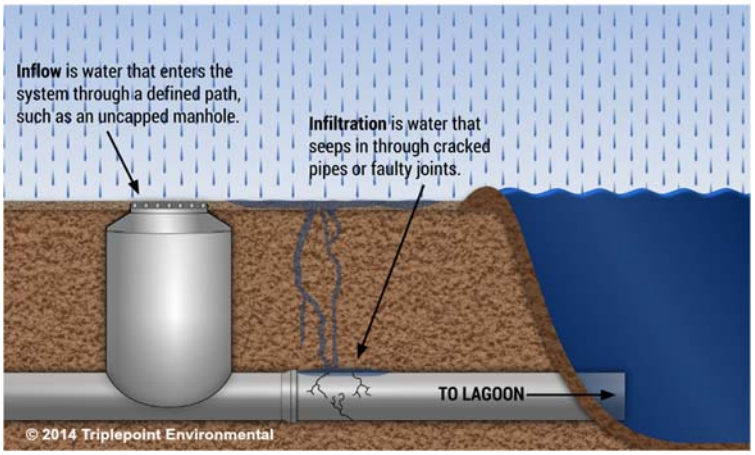

Figure 5. Infiltration and inflow into lagoon systems. Photo courtesy of Triplepoint Technologies (2014). www.triplepointwater.com

There are several ways to reduce I&I into wastewater collection systems. An I&I study typically consists of flow monitoring of the collection system and influent and potentially hydraulic modeling to determine the location and magnitude of I&I in the system. Monitoring during wet weather conditions is important to define the magnitude if inflow. Infiltration can be identified through knowledge of area ground water conditions and locations of sewer pipes that may be below the ground water table. I&I may also be occurring at the lagoon site itself through defections in the lagoon dike or liner.

Issues Addressed

Infiltration and inflow can cause several problems for lagoons, including the following issues:

- Capacity Exceedance/Hydraulic Overload,
- Lower DO,
- Retention time reduction,
- Short-circuiting, and
- Increased treatment costs.

Reduction of I&I can improve overall lagoon operating conditions, not only improving removal efficiency but also reducing overall treatment volumes.

Design Variations

Reduction of I&I can be achieved through multiple technologies; however, selection of the appropriate technology depends upon the type, location, and magnitude of flow contribution which is typically identified through some type of I&I study. Montana does not have any regulatory requirements related to I&I or flow reduction; however, EPA does have guidelines that define excessive infiltration and conditions that require investigation (40 CFR 35.2005, 40 CFR 35.2120) under federal grant assistance projects. Under these guidelines, sewer flows higher than 120 gallons per capita per day (gpcd) require investigation into I&I sources. Through its Water Infrastructure Outreach, EPA Region 1 (EPA 2014a and 2014b) also recently developed a series of guidance documents for

estimating and addressing I&I as well as private sewer laterals, which provide information and guidance for wastewater system operators.

There are many references and prior studies available that describe general I&I investigation and study tactics in varying detail. For example, King County, Washington (2005) has developed an *Executive's Regional Infiltration and Inflow Control Program* through its Waste Water Treatment Division, which, although is specific to the 34 King County wastewater districts, provides valuable information on I&I background, investigation techniques, and cost-benefit analysis, and control recommendations. The Commonwealth of Massachusetts (1993) has developed a document titled *Guidelines for Performing Infiltration/Inflow Analysis and Sewer System Evaluation Survey*.

Inspection Techniques. The following techniques are commonly used as a key part of any I&I study:

- *Dye testing:* A colorful, non-toxic dye is added to suspected infiltration or inflow points and the sewer line is monitored through visual or video inspection to determine whether it is connected to the sanitary system.
- *Smoke testing:* Suspected I&I reaches are isolated with plugs and non-toxic smoke is blown into the reach. Illicit connections, drains, or other defective pipe conditions are identified through the presence of smoke. Smoke testing is not effective for identification of ground water infiltration.
- *Video or Closed Circuit Television (CCTV):* CCTV inspection consists of sending a remote camera through pipe reaches in order to inspect the condition of the pipes. Illicit connections can be easily identified as well as ground water infiltration through defects in the pipe.
- *Physical Inspection:* Physical inspection of sanitary sewer infrastructure can help identify sources of I&I. Inspections of pipe, manholes, lagoon dikes and liners, and other infrastructure should be conducted.
- *Rainfall and Flow Monitoring:* Monitoring of sewer flows throughout the collection system is important in the ability isolate I&I source areas. Rainfall and wet/dry weather monitoring will provide the ability to correlate sewer flows with precipitation and seasonal ground water levels to identify I&I.

I&I Reduction Technologies. The following technologies are commonly used to eliminate I&I in wastewater collection systems:

- *Pipe Replacement:* Pipe reaches with high levels of I&I can be replaced through traditional "dig-and-replace methods", where the old pipe is removed and a new pipe is replaced along the same alignment. In Montana, new sewer mains are regulated under circular DEQ-2 Design Standards for Public Sewage Systems (MDEQ 2012).
- *Pipe Rehabilitation:* As an alternative to pipe replacement, several pipe rehabilitation methods are readily available to eliminate I&I problems.

Per the Collection Systems O&M fact Sheet, Trenchless Sewer Rehabilitation (EPA 1999), Trenchless sewer rehabilitation techniques offer a method of correcting pipe deficiencies that requires less restoration and causes less disturbance and environmental degradation than the traditional dig-and-replace method. Trenchless sewer rehabilitation methods include: pipe bursting/in-line expansion; slip lining; Cured-In-Place Pipe (CIPP); and modified cross-section liner. The applicability of each technology varies with size, type, and condition of the host pipe.

I&I reduction techniques for other types of wastewater infrastructure (manholes, pump stations, lagoons, etc.) require site-specific analysis and repair or replacement recommendations. In addition, removal of direct inflow connections such as roof drains or basement sumps may result in I&I reduction.

Operations and Maintenance Issues

Reduction of non-sewage influent will most likely reduce operation and maintenance requirements at facultative lagoons. Elimination of inflow will reduce overall peak flows into the system wet weather and reduction of infiltration will lower peak flows during dry weather conditions. Influent water quality conditions will be more consistent which will improve the overall performance of the lagoon.

Relative Capital Costs

EPA guidelines (YEAR) identify flows in excess of 120 gpcd as having potential I&I impacts that should be investigation. However, cost-benefit analysis of I&I reduction efforts are difficult to quantify and should be evaluated on a case-by-case basis to determine whether it is more effective to remove the source of I&I or simply convey and treat it at the lagoon.

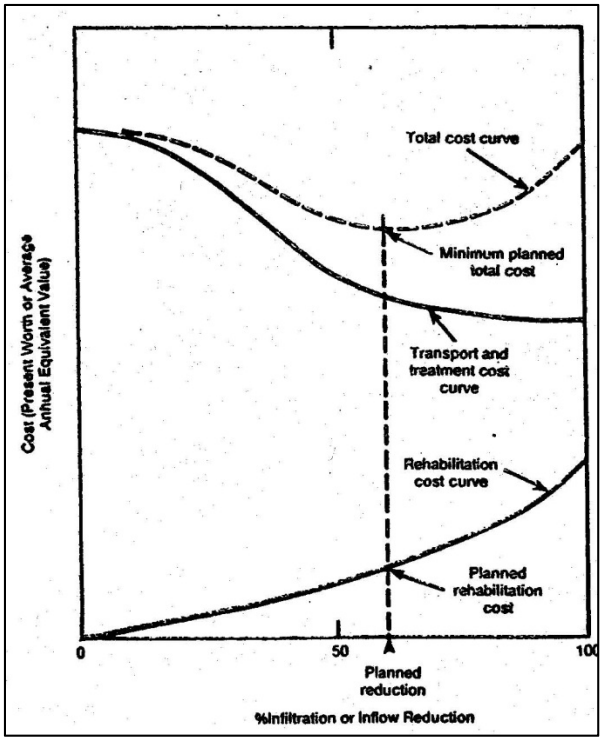

Figure 6. Courtesy of I/I Analysis and Project Certification (EPA 1985)

Costs for different types of I&I reduction techniques are included below.

Dig-And Replace. Dig and replacement costs for sewer main upgrades are variable based upon depth, diameter, soils, and location; but generally per foot costs range from $75 to $150 per foot.

Trenchless Rehabilitation. General per foot costs for different trenchless pipe rehabilitation technologies are included in Table 3.

Table 3. Typical Rehabilitation Cost Ranges for Small Sewer Mains

Technique	Pipe Diameter (in)	Cost Range per linear ft
Pipe bursting	8	$40-$80
Slip lining	21	$80-$170
CIPP	8	$25-$65
Modified Cross Section	8	$18-$50

Source: Collection Systems Fact Sheet-Trenchless Sewer Rehabilitation, EPA 1999.

Private Lateral Replacements and Sump/Foundation Drains.
Lateral Replacements: $5,000 - $8,000
Disconnection of Sump Pump (to Backyard): $500 - $2000

If determined a major source of I&I, replacement of private infrastructure presents some difficulties to wastewater operators, as it presents challenges related to municipal authority and private property rights. Often times, a

leaking lateral line that may be contributing a large amount of flow to the municipal system are not causing any problem to the homeowner. Communities can choose to absorb the costs of replacing lateral lines if determined a high enough priority, or they can offer different types of financing or incentive programs to convince homeowners to upgrade their lines.

Constructability

I&I reduction technologies typically involve rehabilitation or replacement of collection system piping and manholes. These technologies are used on nearly all municipal collection systems; as a result, constructability is relatively straightforward and availability of contractors is good. Some pipe rehabilitation technologies require specialized equipment and supplies; however, there are multiple firms that operate in the Montana region.

Replacement and upgrades to private lateral connections, roof, drains, and foundation drains can sometime present relocation challenges; however, construction methods are well proven.

New/Innovative Approaches

Similar I&I reduction techniques have been used for a long time, and new/innovative approaches are limited. Modern construction equipment and monitoring techniques have increased the reliability and accuracy of I&I studies.

Recommendations

Facultative lagoon operators that are having problems with hydraulic overloading during wet weather events or suspect that significant infiltration is occurring due to deteriorated pipes or shallow ground water should conduct an I&I study. Most I&I enters the wastewater stream through the collection system, although lagoons do have the potential to allow inflow. Various proven I&I reduction techniques and approaches can be implemented to reduce non-sewage inflows to the wastewater system. Replacement of leaking infrastructure and several slip lining technologies are available that are effective in reducing non-sewage influent; these technologies are regularly used throughout Montana. However, depending on the magnitude of I&I, the cost of influent reduction may exceed the resulting additional cost of treatment. Site specific analysis should be conducted.

References

Commonwealth of Massachusetts, Department of Environmental Protection. 1993. Guidelines for Performing Infiltration/Inflow Analysis and Sewer System Evaluation Survey.

Crane S, GHD Limited. Cost-effective and Reliable Inflow-Infiltration Reduction - Have They Got It Right Down-Under?

Metropolitan Council. 2009. Publication #32-09-046. Inflow and Infiltration Toolbox. Available at:

http://www.metrocouncil.org/METC/files/b2/b2ac562a-e78b-4741-b683-542c53852e94.pdf

Montana Department of Environmental Quality, Final October 2012. Circular DEQ-2, Design Standards for Public Sewage Systems.

National Clay Pipe Institute, 1996. Technical Note (TN05), Infiltration and Inflow Studies. Available at:
http://www.ncpi.org/files/TechNotes/TechNote5.pdf

National Small Flows Clearinghouse, Pipeline, spring 1999, Volume 10, No. 2. Infiltration and Inflow can be Costly for Communities. Available at:
http://www.nesc.wvu.edu/pdf/WW/publications/pipline/PL_SP99.pdf

Triplepoint Water Technologies. 2014. Triplepoint's Lagoon Blog. Lagoon Infiltration and Inflow: Problems and Solutions. Available at:
http://www.triplepointwater.com/lagoon-infiltration-and-inflow/#.VS528PnF-So

United States Environmental Protection Agency, Region 1 (EPA). 2014a. Water Infrastructure Outreach, Guide for Estimating Infiltration and Inflow. Available at:
http://www.epa.gov/region1/sso/pdfs/Guide4EstimatingInfiltrationInflow.pdf

United States Environmental Protection Agency, Region 1 (EPA). 2014b. Water Infrastructure Outreach, Private Sewer Laterals. Available at:
http://www.epa.gov/region1/sso/pdfs/PrivateSewerLaterals.pdf

United States Environmental Protection Agency, Office of Water, September 1999. Document 832-F-99-032, Collection Systems O&M Fact Sheet, Trenchless Sewer Rehabilitation.

United States Environmental Protection Agency, Office of Water, September 1999. Document 832-F-99-035, Combined Sewer Overflow Technology Fact Sheet, Inflow Reduction.

United States Environmental Protection Agency, Office of Pollution Control, 1985. Infiltration/Inflow, I/I Analysis and Project Certification, Available at:
https://fortress.wa.gov/ecy/publications/publications/9703.pdf

United States Environmental Protection Agency, 1984 (Accessed via e-CFR on April 16, 2015). 40 CFR, Section 35, Subpart I – Grants for Construction of Treatment Works. 35.2005 & 35.2120.

4.1.1.4 Mosquitos/Insects

General Description

Propagation of mosquitoes and flies due to wastewater treatment ponds may create not only health hazards, but also nuisance conditions. Mosquitoes are

considered the most problematic because they are known as vectors of several diseases. It may be necessary for wastewater lagoons in Montana to develop a basic monitoring and control program for mosquitos and other insects.

Biological Background. Ever since the expedition of Lewis and Clark (1804-1806), the mosquito has been recorded as the worst insect pest of the prairies (Mail 1934). Mosquitos are a group of flies that occur in almost every region of the world except Antarctica. They have been discovered in mines nearly a mile below the surface, and on mountain peaks at 14,000 feet (Flathead City-County Health Department 2015). There are 3,200 recognized species worldwide with approximately 50 species found in Montana (Rolston and Johnson 2012). Mosquitos are the most important arthropod affecting human health due to annoyance, discomfort from bites, and transmission of diseases. They are second to ticks in affecting the health and well-being of livestock and companion animals (Rolston and Johnson 2012). Standing-water mosquitoes (*Culex*, *Culiseta* and *Anopheles* species) overwinter as adults in animal burrows, inside buildings, under bridges, in culverts, etc. (Rolston and Johnson 2012). In late spring they come out of hibernation and deposit eggs on the surface of permanent or temporary pools of water, and larvae hatch from floating rafts of eggs on the surface after a 24- to 48-hour incubation period. The larvae are aquatic and must have non-moving water to complete development through four moltings into the pupal stage, which is also aquatic. Development from egg to adult takes from 7 to 14 days in the summer (Rolston and Johnson 2012).

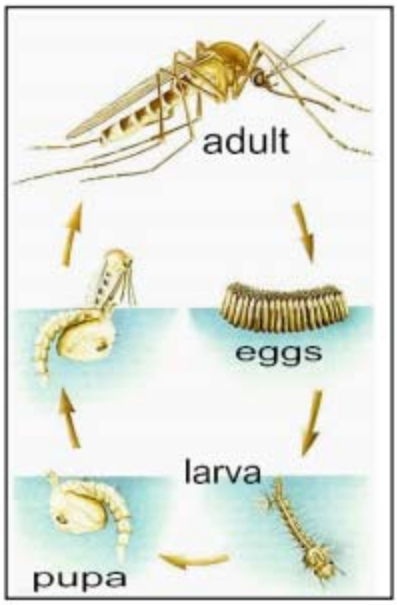

Figure 7. Mosquito life cycle showing the four different life stages.

Shortly after emergence from pupal stage, both male and female mosquitoes feed on floral nectar for fluids and energy, after which they mate and the females search for a suitable animal to bite in order to obtain a blood meal that

is used for the development of eggs (Rolston and Johnson 2012). Females survive an average of 6 weeks, while those with an adequate food supply can live up to 5 months, locating food sources by a combination of the carbon dioxide and other trace chemicals exhaled, and the temperature patterns that their prey produce (Flathead City-County Health Department 2015). Mosquitos have an ecological benefit in serving as food for other organisms; however, this value is somewhat overshadowed by the intense irritation they cause livestock, wildlife, and humans. Aside from the nuisance they cause, mosquitos are a danger because of the several encephalitic viruses and one nematode (canine heartworm) they can transmit. The flight range of an average female mosquito is normally between 1 and 10 miles, but some species can travel up to 40 miles before taking a blood meal and laying eggs (Flathead City-County Health Department 2015). Buffer zones between human developments and adjacent mosquito habitation sites have been recommended by public health officials, but a buffer zone sufficient to avoid contact may not be possible.

Monitoring Mosquito Populations. Mosquito abundance in wetlands and artificial lagoons can be determined using a variety of sampling techniques. Sampling immature mosquito production can identify problem areas and help determine where certain best management practices can be employed to reduce future production. The immature (aquatic) life stages of larvae and pupae are typically sampled using a 14-ounce white "dipper" cup attached to a long wooden or metal handle to determine the relative density of larvae (Kwasny et al. 2004). Mosquito number and species are recorded, resulting in a "number per dip" index (Kwasny et al. 2004). Some species of mosquitos are difficult to sample in their immature stage when aquatic vegetation is abundant, in which case larval traps or the more successful column sampling devices can be used (Kwasny et al. 2004).

The relative abundance of adult mosquitoes (flying stage) is estimated using light traps, carbon dioxide-baited traps, or landing counts. Light trap contents are collected weekly during the mosquito season and the total number of adult females, by species, is divided by the number of active trap nights to determine the index value of adult females per trap night (Kwasny et al. 2004). Landing counts are conducted by an observer facing away from the wind at a sample site and counting the number of mosquitoes that land on her pant legs during a one-minute period, then dividing the count by two for a "leg count" index (Kwasny et al. 2004). This sample is usually done early in the morning or at dusk when temperatures are cool and mosquitoes are most active, and repeated three or four times per site (Kwasny et al. 2004).

Mosquito Control. Weather is assumed to be the most important determining factor in mosquito abundance; the amount of precipitation in conjunction with the temperature determines whether or not mosquitos will reach pest proportions in a region (Mail 1934). Poor water quality tends to increase the production of mosquitoes because the high levels of organic matter and

nutrients (particularly reduced forms of nitrogen such as ammonia) are thought to provide nutrients for the bacteria and algae used as food by mosquito larvae (Walton 2003). Biological controls such as various species of predatory fish are used in certain parts of the country, but wastewater lagoons specifically and the Montana climate in general does not have conditions that can support fish. A minnow called the mosquitofish (*Gambusia affinis*) was successfully used as a biological control in California, Spain, Italy, and other countries before it was shipped to Banff, Alberta for experimentation in the pools of the district, only to find that it could not propagate outside of the warm sulphur springs (Mail 1934). Frogs, salamanders, and other amphibians also prey upon mosquitos in Montana, but wastewater lagoons do not provide adequate habitat for them to function as control factors. Specifically in the case of wastewater treatment, the decomposition of organic matter and conversion of ammonium to other forms of nitrogen in the nitrogen cycle require considerable amounts of oxygen, which can lead to low DO concentration and can create unsuitable conditions for aquatic mosquito predators such as predatory insects and fish (Walton 2003). Birds and bats, natural predators of immature and adult mosquitos, constitute a check on the multiplication of mosquitoes but not large enough to control the proliferation of the pest. Historically, "oiling" of the water surface using material from an oil refinery or automobile crankcase drainings was considered an efficient means of mosquito control (Mail 1934), but now mineral oils and monomolecular films that disperse as a thin layer on the water surface are used to cause larvae and pupae to drown (EPA 2015). This older remedy can be damaging to aquatic plants and other beneficial aquatic insects, and is only recommended as a last resort. The benefit of larvicidal films is that mosquitoes cannot develop a resistance, since control is through a physical mode of action. Creosote was also historically used as a possible deterrent of adult mosquitos by spreading it on the doors of livestock enclosures (Mail 1934), but this substance is now a restricted-use pesticide known to contain hazardous chemicals and is a possible human carcinogen (ATSDR 2002; EPA 2007).

Issues Addressed

> *"The amount of total Kjeldahl nitrogen (TKN), the chemical oxygen demand (COD), and total organic carbon (TOC) will dramatically affect the suitability of a waste lagoon for the larvae of Cx. quinquefasciatus. Total Kjeldahl nitrogen values for lagoon water of less than 50 mg per liter or greater than 500 mg per liter (4.2 pounds per 1,000 gallons) kills early stage mosquito larvae. TOC values that fall outside the range of 100 to 1,000 mg per liter are also effective, as are COD values of less than 400 mg per liter and greater than 2,000 mg per liter. However, effluent values that are in the appropriate range for mosquito control may not be effective if bank vegetation and floating debris are neglected.*

"Total nitrogen appears to be the most important characteristic of lagoon water when it comes to controlling mosquitoes. High TKN values alone will dramatically reduce larval survival even when TOC and COD values are ideal for mosquito production.

"Keep in mind that there is a considerable amount of variation between lagoons. Also keep in mind that adverse weather will affect a lagoon's nitrogen content. Prolonged periods of heavy rain may dilute nitrogen levels that otherwise would be high enough to retard mosquito breeding. Dilution can be beneficial as well. If lagoon nitrogen is relatively low, heavy rains may further reduce levels to a point below the minimum needed for mosquito breeding. Drought will have the opposite effect by raising the concentration of nitrogen as lagoon water evaporates. Such changes are often temporary, but may last long enough to allow at least one generation of mosquitoes to emerge. Testing is the only way to be sure a lagoon's nitrogen values are high or low enough to discourage mosquito breeding. It is advisable to test lagoon water in May or June to evaluate its suitability for mosquito production" (Stringham and Watson 2002).

Design Variations

Modern-day mosquito control includes low toxicity larvicides and adulticide chemicals that are toxic to bees, fish, and aquatics if not used in accordance with the labels.

Adulticides. In Flathead County Mosquito Control, fogging with adulticide chemicals from a truck-mounted machine has an effective spray drift of about 300 feet and is conducted in the evening when bees have returned to their nests or hives (Flathead City-County Health Department 2015). The most common material used in low-volume spraying is pyrethrins, a naturally occurring substance harvested from chrysanthemums that degrades within 4 to 6 hours after spraying; and pyrethroids, a synthetic form that is also commonly used (Placer Mosquito and Vector Control District 2015). Organophosphate insecticides affect the nervous system of insects, but frequent use can lead to pesticide resistance in the mosquito population, so these insecticides are used in rotation with pyrethrins and pyrethroids to help ensure that adulticide applications remain effective (Placer Mosquito and Vector Control District 2015). In concentrated areas, insect electrocutors (bug zappers) and mosquito trapping devices can be used to attract, repel, or kill outdoor infestations. Some of these devices are designed to mimic a potential mammalian host (horse, cattle, man, and domestic pets) by emitting a plume of carbon dioxide, heat, and moisture, then using a vacuum device to suck the insects into a net or cylinder where they dehydrate and die (AMCA 2015). Scientific data on the effectiveness of these devices is sparse, and some mosquito traps can be quite expensive. Due to a higher level of environmental and human health risk compared with natural mosquito control methods, chemical controls should be

seen as a last resort and are best left to certified pesticide applicators (Ladd and Frankenberger 2003). Reduction of breeding sources for mosquitos is another major method of control, but unfortunately wastewater lagoons can act as a major source for mosquito production. Surrounding vegetation could be managed to reduce the number of areas where adult mosquitoes can find shelter. This can be done by mowing, controlled burning disking (to reduce dense stands of emergent or undesirable vegetation, and provide favorable conditions for the establishment of most-soil plants consumed by waterfowl), and haying and grazing (which has little available information on the effectiveness as a mosquito control measure) (Kwasny et al. 2004).

Larvicides. Liquid larvicide products are applied directly to water using backpack sprayers and truck or aircraft-mounted sprayers (EPA 2015). Tablet, pellet, granular, and briquette formulations or larvicides are also applied by mosquito controllers to breeding areas (EPA 2015). Commonly used larvicides include bacterial insecticides such as *Bti* (*Bacillus thuringiensis israelensis*) and *Bs* (*Bacillus sphaericus*). *Bti* is a naturally occurring soil bacterium that produces proteins in a crystalline form that causes mosquito larvae that eat these crystals to have trouble digesting food and die of starvation. *Bti* is quickly biodegraded and leaves no residue, and is not harmful to fish, pets, wildlife, or humans. *Bacillus sphaericus* is also a common soil-inhabiting bacterium, and produces a protein toxin that may be used to control mosquito larvae. *Bs* is commonly used to control mosquito larvae in highly polluted water, such as sewage treatment plants. *Bs* is nontoxic to non-target organisms, although some natural recycling of this organism is likely and products usually last for about 21 days (AMCA 2015). Another larvicide method is Methoprene, an insect growth inhibitor that mimics a natural juvenile hormone, and when present in the larval habitat keeps immature insects from maturing into adults and causes them to die in the pupal stage (Placer Mosquito and Vector Control District 2015).

Regular ponds with mosquitos can utilize gambusia, known as mosquito fish, which are similar to a guppy and capable of consuming over two hundred mosquito larvae in an hour. Unfortunately, they will also attack dragonfly larvae, which are another predator to mosquito larvae. They naturally live in the southern United States, but some varieties have been bred to survive colder temperatures, even down to -30°F. Ponds with good aeration pumps generally do not produce mosquitos because the agitation of the water surface interferes with the female mosquito's ability to lay eggs. Vegetation on and around the pond surface can potentially provide shelter for mosquito larvae, which can be a problem in wastewater lagoons since a biofilm is a beneficial component of treatment. Recently, the LarvaSonic method of larval control has become available, which is a system that transmits sound energy into water at the resonant frequency of the mosquito larvae air bladders and instantly ruptures the internal tissue, causing death (AMCA 2015). Finally, water deeper than 5

feet, with a maintained flow that disperses water, or with artificial or wind-driven aeration creates conditions that are less conducive for mosquito production (Walton 2003). Waves and water disturbance in open water zones are not favorable to mosquito egg laying and can drown immature mosquitoes (Walton 2003).

Operations and Maintenance Issues

Maintaining low rates of mosquito production is likely to depend on periodic use of mosquito control agents. Regular monitoring and sampling of immature and mature mosquito populations should be implemented prior to remediation to determine the level of need. Light trap contents are collected weekly during the mosquito season. Landing counts of adults should be done early in the morning or at dusk when temperatures are cool and most active, and repeated three or four times per site (Kwasny et al. 2004). Vegetation management and regular removal or floating debris is necessary to reduce cover for all stages of mosquitoes: heavy, unmowed vegetation that hangs into the water along a lagoon's shoreline provides additional cover for mosquito larvae, and surrounding vegetation provides shelter for adult mosquitoes (Stringham and Watson 2002). Weekly or biweekly mowing will eliminate excessive vegetation and reduce the volume of clipped plant material left floating in the water after mowing. Mosquito control is moving away from application of conventional chemical pesticides to more environmentally friendly methods such as mosquito-specific bacteria in the genus *Bacillus* and insect growth regulators. *Bacillus sphaericus* (*Bs*) is a more effective control agent for mosquitoes in wastewater with high organic content and/or suspended sediment than *Bacillus thuringiensis israelensis* (*Bti*) (Knight et al. 2003). However, *Bs* has a narrower host range than *Bti*; most *Aedes* species are not susceptible to *Bs* (Knight et al. 2003). The effectiveness of the toxins against mosquito larvae will vary with environmental conditions such as the concentration of suspended solids, water temperature, water depth, ionic content of the water, larval density, solar radiation, flow regime, and vegetative cover (Walton and Mulla 1992). Spores containing the *B. sphaericus* toxin precursors readily settle out of the water column so repeated applications of a lagoon would be necessary, although *Bs* should be used in a controlled manner to reduce the likelihood of resistance induction in the mosquitos. The effectiveness of control measures for immature mosquitoes declines in thick vegetation and with poor water quality. In shallow, standing water, mosquito-specific microbial insecticides can be effective for longer than 1 week; however, dilution of mosquito control agents can be significant in lagoons due to their short water residence times, so more frequent application will be necessary and must be determined based on sampling. Integrated pest management approaches combine source reduction, biological control agents, and application of mosquito-specific larvicides. The volume and flow rate of wastewater treatment lagoons may make adulticiding a necessary means of controlling mosquitoes, but should be used primarily as a last resort to control mosquitoes. For situations where larviciding is ineffective or under

emergency situations when a disease outbreak is imminent, adulticiding is the only effective means of quickly eliminating infected mosquito populations (Knight et al. 2003). Recent studies indicated that adulticides (i.e. pyrethrin, malathion, and permethrin) applied at mosquitocidal dosages were not acutely toxic to common freshwater insects and aquatic vertebrates (Lawler et al. 1997). Even though the insecticides currently used to control adult mosquitos are safe, this method of control is not universally accepted by the general public because of perceived negative impacts (Walton 2003). Operations personnel should follow all warnings and precautions on mosquito control supplies, or hire a professional to conduct the control applications.

Relative Capital Costs

Control measures against the stages of the mosquito life cycle that occur in water are preferred to measures directed against adult mosquitoes. After mosquitos become adults capable of dispersing from the wetland, control becomes more difficult, costly, and requires use of insecticide across a comparatively broad geographic area (Walton 2003). The cost of mosquito abatement for small wetlands is several hundred dollars for each application of mosquitocidal materials (Knight et al. 2003). The repeated use of treatments would be necessary in lagoons with less retention time. Using several abatement strategies simultaneously when monitoring determines that an outbreak is occurring or imminent would compound the cost but be more effective and hopefully prevent repeated, large, infestation events. The appropriate use of insecticides will bring an infestation under control within 1 to 3 days (Stringham and Watson 2002). The Larvasonic device sells for $4,000 per unit (estimates from 2002, and does not include the reflector to extend the range or the labor cost of frequent applications).

Constructability

Monitoring can be easily conducted by trained personnel at the lagoon site. Sampling of immature mosquitoes should be done regularly to identify potential problem areas where remedial action should be applied. Sampling of mature mosquitos is uncomplicated; traps are hung from trees, buildings, or special stands and usually remain in the same place throughout the season and from year to year (Kwasny et al. 2004). Preventative methods such as control of surrounding vegetation would be simple and should be attempted first to reduce shelter sites for adult and immature mosquitoes. Industry standards on larviciding should be strictly followed, but the application of the pellet forms of *Bs* and *Bti* are relatively easy. If targeting of adult mosquitoes is deemed necessary, adulticides should be applied at mosquitocidal dosages.

New/Innovative Approaches

The LarvaSonic method of larval control has become available, which is a system that transmits sound energy into water at the resonant frequency of the mosquito larvae air bladders and instantly ruptures the internal tissue, causing death (AMCA 2015). The device transmits acoustic energy between 10 kilohertz

and 36 kilohertz, just above the level that humans can hear (Schuessler 2002). Larvasonic is an effective tool for mosquito larval control where pesticides could cause damage to nearby sensitive areas. The company that markets the technology, New Mountain Innovations in Niantic, Connecticut, claims that larvae within 3 feet of the device in the water column will die instantly, and larvae within 5 feet will die within 24 hours (Schuessler, 2002). A separate study found an average mortality rate of 59 to 84% in standing water sites, with many of the effects appearing to increase over time (7 to 14 days) (Taylor 2015). However, they acknowledged that the sampling of treated mosquito larvae could be underestimating the number of affected larvae because they died and sank to the bottom instead of being available in the water column for capture and testing. In large areas, a "reflector" must be added to the device to increase the range, and frequent visits are required to continue the treatment because there is no residual effect for newly-deposited larvae (Taylor 2015). This is a relatively new product and the frequent applications may make it undesirable as the primary control method for lagoons. The original model only had a kill range of 3 to 5 feet, but a "Field Arm" mobile wetlands unit claims a kill range of 15 feet, the "Canal Pod" towed unit claims a 50 foot instant kill radius, and industrial units with custom installation are also available for large scale mosquito problems (New Mountain Innovations 2015).

Recommendations

Although mosquito control is not directly related to reduction of ammonia, total nitrogen, and total phosphorus, lagoon operators should budget for preventive measures such as periodic removal of surrounding vegetation, and regular monitoring of immature and mature stages of mosquitoes. Application of several abatement strategies would be most effective. Higher flow rates, perhaps facilitated by a longer, sinuous flow patterns created across the lagoon with the use of floating barriers/baffles would make laying of mosquito eggs more difficult, as would artificial aeration, if possible for the type of lagoon used at a particular site. Larvicidal treatments should be attempted before adulticidal controls, and methods should be carefully alternated to prevent resistance induction in the mosquitos.

References

AMCA. 2015. The American Mosquito Control Association; University of Florida. Available at: http://www.mosquito.org/control. Accessed April 16, 2015.

ATSDR. 2002. Toxic Substances Portal- Creosote; Public Health Statement for Creosote. Agency for Toxic Substances & Disease Registry. Available at: http://www.atsdr.cdc.gov/PHS/PHS.asp?id=64&tid=18. Accessed April 16, 2015.

EPA. 2007. Preliminary Risk Assessment for Creosote. U.S. Environmental Protection Agency. Available at:

http://www.epa.gov/pesticides/factsheets/chemicals/creosote_prelim_risk_assess.htm. Accessed April 16, 2015.

EPA. 2015. Controlling Mosquitos at the Larval Stage. Available at: http://www2.epa.gov/mosquitocontrol/controlling-mosquitoes-larval-stage. Accessed April 16, 2015.

Flathead City-County Health Department. 2015. Flathead County Mosquito Control. Available at: http://flatheadhealth.org/mosquito-control/. Accessed April 16, 2015.

Knight, R.I., W.A. Walton, G.F. O'Meara, W.K. Reisen, R. Wass. 2003. Strategies for effective mosquito control in constructed treatment wetlands. Ecological Engineering, 21, pp 211-232.

Kwasny, D.C., M. Wolder, C.R. Isola. 2004. Technical guide to best management practices for mosquito control in managed wetlands. Central Valley Joint Venture: California Department of Fish and Game, U.S. Fish and Wildlife Service, Sacramento, CA.

Ladd, B. and Frankenberger, J. 2003. Management of Ponds, Wetlands, and Other Water Reservoirs to Minimize Mosquitoes. Water Quality: The Purdue Extension Water Quality Team, Purdue University. WQ-42-W, June 2003. Available at: https://engineering.purdue.edu/SafeWater/Ponds/WQ-41-W.pdf. Accessed April 16, 2015.

Lawler, S.P., Jensen, T., Dritz, D.A., 1997. Non-target effects of mosquito larvicides used on national wildlife refuges. University of California Mosquito Control Research Annual Report. University of California, Berkeley, CA, pp. 34–35.

Mail, G.A. 1934. The Mosquitoes of Montana. Bulletin No. 288, March 1934. Montana State College, Agricultural Experiment Station, Bozeman, Montana.

New Mountain Innovations. 2015. LarvaSonic. Available at: http://www.larvasonic.com/applications.html. Accessed April 17, 2015.

Placer Mosquito and Vector Control District. 2015. http://www.placermosquito.org/

Rolston, M. and Johnson, G. 2012. Biology and Impact of Mosquitos in Montana. Montana State University Extension, MT201204AG. Available at: http://store.msuextension.org/publications/AgandNaturalResources/MT201204AG.pdf. Accessed April 16, 2015.

Schuessler, H. 2002. Going after mosquitoes, with a racket. The New York Times. Published: June 13, 2002. Available at: http://www.nytimes.com/2002/06/13/technology/going-after-mosquitoes-with-a-racket.html. Accessed April 17, 2015.

Stringham, M. and W. Watson. 2002. Managing waste lagoons to control mosquito breeding. N.C. State University, Department of Entomology. Available at: http://www.ces.ncsu.edu/depts/ent/notes/MedVet/wnv-manage.htm. Accessed April 17, 2015.

Taylor, K. 2004. Ultrasound as a pest management tool for WNv: An efficacy study of mosquito larvae in catch basins and standing water in Guelph, ON. Presentation available at: http://www.ovca.ca/Day%202/PDF/OVCA_con_day2_115.pdf. Accessed April 17, 2015.

Walton, W.E. and M.S. Mulla. 1992. Impact and fates of microbial pest-control agents in the aquatic environment. In: Rosenfield, A., Mann, R. (Eds.), Dispersal of Living Organisms into Aquatic Ecosystems. Maryland Sea Grant College, University of Maryland, College Park, MD, pp. 205–237.

Walton, W. 2003. Managing Mosquitoes in Surface-Flow Constructed Treatment Wetlands. UCANR Publications; University of California, Division of Agriculture and Natural Resources.

4.1.1.5 Maintenance Schedules/Activities

General Description

Many documents regarding maintenance schedule are available, most focused on sludge accumulation, vegetation (both bank and aquatic) control, algae control, burrowing animal control and maintenance of embankments. Many references researched included similar information. All articles support that properly maintaining facultative lagoons results in lower overall operation costs and improved wastewater treatment.

Issues Addressed

Proper operation and maintenance of lagoon systems results in reduced costs and increased nutrient removal (FCMNRC 2004).

Lagoons are intended to accumulate sludge, which is accelerated in colder climates but the decay of the sludge is reduced as well. Sludge accumulation becomes an issue once the sludge storage volume has been exceeded. To eliminate discharge of sludge proper maintenance of the sludge volume must be addressed (FCMNRC 2004).

Short circuiting is the most common cause of lagoon performance issues. Optimizing the system by improving flow patterns can significantly improve performance (FCMNRC 2004).

Control of both terrestrial and aquatic vegetation is critical for structural stability and improved nutrient removal. Burrowing animals can compromise embankments and lead to seepage and possible embankment failure.

Design Variations

Optimization and expansion of an existing lagoon is generally less expensive than replacement with mechanical treatment plants due to reduced energy costs and reduced operational challenges (FCMNRC 2004). Optimizing lagoon systems can be achieved by operational and design changes (improve flow patterns), flow scheme modification (multi cell lagoons), mechanical aeration, chemical addition (enhancing settling and Phosphorous removal), pre and post treatment processes (reduce lagoon loading and improving effluent quality), and reduced odor emissions (FCMNRC 2004).

Optimization. System performance can be significantly improved through optimization to eliminate short circuiting and dead spaces which can be indicated with dye tests (FCMNRC 2004).

Operational and design changes can improve system performance. This can be achieved by (FCMNRC 2004):

1. Installing baffles at the lagoon inlet and outlet;
2. Inlet and outlet relocation;
3. Recirculating flow from outlet to inlet for improved mixing;
4. Adding inlets and outlets;
5. Operating multiple cells in parallel instead of series;
6. Removing vegetation and solids that affect flow patterns;
7. Installing or relocating mixers and aerators; and
8. Relocating the discharge invert to avoid discharge of algae and solids/sludge.

Modifying the flow timing can greatly improve effluent quality, especially during cold months and pond turnover periods. Timing discharges to when algae levels are lower produces higher discharge water quality. Unfortunately these are the periods of higher flow, therefore retention times must be met for the best effluent quality. Careful modification of the discharge scheme so that discharge is proportional to receiving water flow rate will avoid impact to receiving water quality (FCMNRC 2004). Ponds operated in series generally produce better quality effluent than in parallel if the first cells are not overloaded. Parallel ponds can be optimized by the addition of baffles to encourage plug flow (FCMNRC 2004).

Lagoons should be designed and operated in such a way that sampling is practical and representative of the overall wastewater effluent quality (FCMNRC 2004). Sampling routines should follow best sampling practices for accurate representation of water quality, this includes sampling location, sample preservation, and sample type and frequency (grab versus composite) (FCMNRC 2004).

Properly operating lagoons must be well maintained. This includes:

1. Eliminating berm damage from rodents and other burrowing animals through trapping or shooting (SME 2004).
2. Eliminating berm seepage by inspecting dyke toe and repairing to avoid failure (SME 2004).
3. Reducing erosion on interior and exterior slopes by installing rip rap (wave action) vegetation maintenance and ensuring proper slope grading (SME 2004). Fast growing grass with a shallow root system that can easily be maintained is desirable to prevent erosion. Other materials such as rock or geosynthetics, can be used from one foot above the high water line to the toe of the dike. These materials can also prevent burrowing animals (H&S Environmental 2003).
4. Removal of sludge mounds prevents dead spots, odors, short circuiting, and plugged inlets and outlets (SME 2004).
5. Maintaining overflow structures to prevent overtopping of berms and structural failure (SME 2004).
6. Repair fences and gates to prevent access from unauthorized personnel and animals.
7. Controlling vegetation is imperative to optimize performance (dead spots and short circuiting) and eliminate mosquito breeding areas. This includes cutting or burning vegetation at the water's edge and cutting or removing vegetation within the cells to improve sunlight penetration for algae growth. Other vegetation that reduces wind mixing action should be maintained as well (SME 2004). Aquatic herbicides should be carefully applied as a spot treatment to keep up on weed control before larger issues are presented. Usually small applications of proper herbicides will not affect wastewater treatment but their use should be approved by governing agencies (H&S Environmental 2003).

Desludging. This operation is relatively infrequent (5 to 20 years) depending on loading rates and overall lagoon size (FCMNRC 2004). Sludge accumulation should be measured annually for desludging planning based on the available storage remaining for sludge. Two common methods for desludging are: 1) mixing the sludge with lagoon water and pumping the mixed water, and 2) draining the lagoon and removing the settled sludge using heavy equipment (FCMNRC 2004). Raft mounted dredges may be used for the mixing and pumping removal. Desludging single cell lagoons is generally more complicated as the cell cannot be taken out of service. In this case sludge is pumped from the bottom with floating raft dredges. Individual cells of multi cell lagoons can be taken out of service for more routine desludging operations. This typically occurs in late fall where the cell is drained so the sludge can concentrate during the colder months for easier desludging in the spring (FCMNRC 2004). During desludging operations it is critical that the lagoon bottom and embankments are not damaged (FCMNRC 2004). Typically removed sludge can be land applied if the lagoon has not received any raw sewage for three months prior to desludging (FCMNRC 2004).

Inspections. Lagoon operators should routinely inspect the lagoons for proper operation and to ensure that maintenance issues are addressed. These inspection items include (H&S Environmental 2003):

- Inspection of pond berms/dykes/levees and pond linings (erosion and leaks),
- Nuisance weeds,
- Insect activity (mosquitos, wasps, etc.),
- Burrowing animals (rodents, muskrats, snakes, etc.),
- Vandalism,
- Mechanical equipment and piping for influent and effluent structures, aerators, etc., and
- Overall system operation indicators: color, odor, algae, foam, dead spots, sludge mounds, temperature, etc.

Algae Control. Excess algae causes many issues, including: nuisance odors, insect activity, clogging of hydraulic structures, reduced mixing, and reduced light penetration. Manually removing algae mats is an effective control method (H&S Environmental 2003).

Maintenance Records. Maintenance activity should be documented and kept on file for inspection by governing agencies. This can include: vegetation control, system modifications, replacement of parts, nuisance animal incidences, vandalism, and equipment inventory (H&S Environmental 2003).

Operations and Maintenance Issues

Activities under this category are all maintenance-related; as a result, they require regular activities on the part of lagoon maintenance staff. Most lagoons are gravity flow; therefore, proper operation of structures must be ensured. Earthen embankment structures must be inspected for damage, erosion, leaking, and rodent damage (EPA 2015). Other O&M issues have been addressed in other sections.

Relative Costs

Proper maintenance and operation of facultative lagoons is generally cheaper than secondary treatment plants (FCMNRC 2014). Routine maintenance of the system is also cheaper than addressing larger issues in the future (H&S Environmental 2003).

Recommendations

There are several well documented maintenance activities and BMPs that should be part of a thorough routine maintenance plan for any facultative lagoon. Completing routine maintenance of the system will help maintain treatment effectiveness and avoid potentially large capital replacement or maintenance in the future. Properly documenting all maintenance activities is critical for cost tracking, regulatory requirements, and proper planning.

References

Federation of Canadian Municipalities and National Research Council (FCMNRC). 2004. Optimization of Lagoon Operation. Available at: https://www.fcm.ca/Documents/reports/Infraguide/Optimization_of_Lagoon_Operations_EN.pdf

H&S Environmental. 2003. Wastewater Lagoon Troubleshooting: An Operators Guide to Solving Problems and Optimizing Wastewater Lagoon Systems.

Saskatchewan Ministry of Environment and The Water Security Agency (SME). 2004. Two Cell Lagoon Operation and Maintenance. Available at: http://www.saskh20.ca/DWBinder/epb310.pdf

U.S. Environmental Protection Agency (EPA). 2015. Facultative Lagoon Wastewater Fact Sheet. Available at: http://water.epa.gov/scitech/wastetech/upload/2002_10_15_mtb_faclagon.pdf

4.1.2 Lagoon Covers

General Description

Very few peer-reviewed documents were found regarding covers for facultative lagoon used for domestic wastewater treatment. Numerous articles were found focusing on animal wastes from feed lots and dairies, but their focus on collecting the bio-gas generated from the concentrated wastes in those lagoons. Also, several vendors had information on their web-sites and promotional documents regarding lagoon covers. In general, the addition of a cover to the lagoon can reduce unwanted variables and capture unwanted emissions. A cover can be a floating plastic cover, a synthetic or natural cover of peat, straw, polystyrene, or high density polyethylene (HDPE) floating balls or hexagonal tiles. The installation of synthetic covers ranges from expensive and laborious to simple and relatively inexpensive, depending on the type of cover and the purpose of the cover.

Issues Addressed

Lagoon covers can influence several factors associated with lagoons and ponds depending on the design and function of the pond. The main issues addressed specifically for facultative lagoons are the following: odor, algae (TSS), heat loss, evaporation, and birds or waterfowl. Depending on what issues need to be addressed, the covers may need to be placed at different locations. For example, if TSS is the main concern in the facultative lagoon, the cover should be placed on the last section of the pond, shortly before the discharge point. This will block the sunlight in that portion of the pond, preventing the growth of algae just before the water is discharged. Algae in certain portions of the pond are beneficial and aid in the nitrification process. Excessive algae growth in shallow settling ponds and large lagoon systems can result in elevated TSS levels and reduced effluent ultraviolet (UV) transmissivity. Blocking sunlight with a cover on the final settling cell prevents new algae growth and causes existing algae to die off and settle to the cell bottom (Nelson Environmental 2015).

Design Variations

Two major types of synthetic covers are currently being used for facultative lagoons: floating membrane covers and hollow floating sphere/floating hexagon shaped tiles. Each is discussed in more detail below.

Floating Membrane Covers: A floating membrane cover, such as the covers manufactured by Lemna Technologies, Inc., may offer thermal protection for facultative lagoons. These covers are typically the most insulating of the covers that will be discussed. By providing additional thermal protection in the lagoons, the water temperature may be warmer, which would help minimize freezing more effectively. The membrane cover provides insulation (R-values ranging from R4 to R16). The access to the lagoon and any equipment in the lagoon would be limited when using a floating membrane cover, which is a significant hindrance for using this product. There are also concerns related to snow loading on the floating membrane cover. Rain and melted snow would easily drain through the spaces between the cover panels; however, accumulated snow on top of the cover could compromise the integrity of the cover as well as its anticipated benefits. The floating membrane cover is therefore determined to be infeasible for Montana facultative lagoons and will not be further evaluated in this report.

Floating Ball/Tile Covers: HDPE ball-type covers are simple, effective methods of covering settling cells. The balls automatically arrange themselves into a closely packed formation covering over 91% of the surface area. The covers can also be used to prevent waterfowl from congregating on lagoons and ponds. Minimizing waterfowl reduces organic loading in the lagoons and bird fatalities in ponds where water may be toxic to wildlife. Ball-type covers are used extensively at or near airport locations to prevent bird fatalities to minimize the potential for aircraft bird strikes (Nelson Environmental 2015).

Some advantages of the ball-type covers include:

- No impediment to aeration equipment pond;
- Prevent UV light penetration and surface oxygenation, maintaining anaerobic conditions;
- Quick and simple to install;
- Spread automatically as the liquid levels rise and fall;
- Virtually maintenance free;
- Virtually indestructible UV resistant HDPE material;
- Cannot tear like conventional membrane covers; and
- Can freeze into ice covered lagoons without damage.

Operations and Maintenance Issues

None of the floating cover options require much maintenance, but the floating ball/tile cover alternatives allow for much easier maintenance of any lagoon equipment present. The floating balls or floating tiles can be pushed aside or temporarily removed from the lagoons as needed, with minimal effort. Maintaining the lagoon equipment with the fixed insulated cover would require removal of a panel or multiple panels of the cover.

Anticipated benefits are based on how well the technology is expected to provide the intended benefits of controlling odors, minimizing algae (TSS), preventing heat loss, limiting evaporation, and eliminating birds or waterfowl from the lagoon. The floating ball/tile cover alternative provides the highest benefits based on the issues associated with facultative lagoons. The floating balls do not provide the level of insulation that the insulated tiles do, but they do provide sufficient insulation to minimize freezing through the lagoons. Based manufacturer information, with an influent wastewater temperature of 50°F and an air temperature of 10°F, heat loss through the lagoons with the floating ball cover would cause the water temperature in the lagoon to equalize at about 39°F.

Relative Capital Costs

The floating ball cover system would be the least expensive option to purchase and install. The total cost of material, delivery, and installation ranges from approximately $2.75 to $3.25 per square foot depending on the location and area needing coverage. Some minor design and coordination with the manufacturer will be required resulting in additional charges, probably less than $5,000 per lagoon. To cover the lagoon with the floating insulated tile cover, the total cost of material, delivery, and installation ranges from approximately $6.15 to $6.90 per square foot. Similar to the floating ball alternative, it is anticipated that minor design and coordination with the manufacturer will cost about $5,000. The estimated useful life for both the floating ball and floating tile options is approximately 20 years.

Constructability

Constructability, for the purposes of this evaluation, includes ease of installation, construction time, and ability to complete construction before the lagoon freezes. The floating tile and floating ball covers are very simple to install, can be installed quickly, and would be able to be installed before the next winter season. Whether tiles or balls are selected to be used in the lagoons, the installation would be the same. The balls or tiles are then shipped to the site and simply poured into the lagoons and will form themselves into a floating cover. The lead time for shipping is anticipated to be about two to four weeks, and installation may be a few days.

New/Innovative Approaches

Floating Solar Panels: Floatovoltaics, or floating solar panels, are an emerging technology that has the potential to be used in facultative lagoons in Montana. In Montana alone, the surface area of wastewater ponds, reservoirs, and sewage treatment pools totals in the thousands of acres. These unused, open spaces are typically hidden from public view and provide an excellent opportunity for floating solar panels.

Photovoltaic systems have been installed on a small number of water bodies in Japan, France, India, the United Kingdom, and the United States. The first system in the United States was recently installed at a vineyard in Napa Valley, California. In an effort to both conserve vine acreage and expand on their existing 1,300-panel system, the winery placed 1,000 solar panels over an irrigation pond. The panels floated on 130 pontoons and were connected to an existing land-based panel system. The 2 systems collectively generated 477 kilowatts at peak output. Installing the panels on the pond, the winery

was able to save almost an acre of vines, equivalent to $150,000 worth of bottled wine annually.

Land conservation aside, floating panels might offer more benefits than ground-mounted solar panel systems. For example, the water underneath provides an effective, natural cooling system. As solar panels continuously absorb sunlight throughout the day, they become very hot. The efficiency of a typical silicon-based solar panel is reduced by about 0.5% for every °C increase over the module's rated temperature (usually around 25°C). Because of water's cooler environment, floating panels do not experience a significant loss in power output when temperatures increase. One study found that solar panels were 8 to 10% more efficient when paired with water.

The addition of a floatovoltaic system to a treatment facility can offset some of these costs and also provide a valuable shading structure. Reduced exposure to sunlight has the potential to decrease evaporation and algae growth in a water body. Evaporation limits water conservation while algae buildup clogs piping and causes other issues, increasing maintenance costs for treatment facilities. The large percentage of underutilized municipal surface water in Montana may present an opportunity for renewable electricity generation.

Recommendations

It is recommended that the floating ball cover be considered for any lagoons that may have issues with TSS (algae) and/or temperature issues. This is the least expensive option, and the floating cover system could be installed very quickly. It is also compatible with other technologies, such as floating barriers.

References

Nelson Environmental. 2015. Available at: http://www.nelsonenvironmental.com/

Advanced Water Treatment Technologies (AWTT). 2015. Available at: http://www.awtti.com/index.php.

Brown, Matt. 2015. Sonoma clean power inks deal for floating solar panel project. The Press Democrat.

Colorado State University Extension. 2012. Best management practices for reducing ammonia emissions: lagoon covers.

DeGarie, C.J., T. Crapper, B. M. Howe, B.F. Burke, and McCarthy. 2000. Floating geomembrane covers for odour control and biogas collection and utilization in municipal lagoons. Water Science and Technology, Vol 42, nos 10-11, pp 291-298.

Environmental Protection Agency, Wastewater technology fact sheet: facultative lagoons.

Funk, T.L., R. Hussey, Y. Zhang, and M. Ellis. 2004a. Synthetic covers for emissions control from earthen-embanked swine lagoons. Part I. Positive pressure lagoon cover. Applied Engineering in Agriculture 20:233-238.

Funk, T.L., A. Mutlu, Y. Zhang, and M. Ellis. 2004b. Synthetic covers for emissions control from earthen embanked swine lagoons. Part II. Negative pressure lagoon cover. Applied Engineering in Agriculture 20:239-242.

Hubbard, R.K., G.J. Gascho, and G.L. Newton. 2004. Use of floating vegetation to remove nutrients from swine lagoon water. Transactions of the ASAE 47(6):1963-1972.

LEMNA Technologies, Inc. 2015. Webpage available at: http://www.lemnatechnologies.com/

Miner, J.R., W. Robarge, D.B. Harris, R. Sheffield, C.M. Williams, F.J. Humenik, J.M. Rice, and D.M.C. Rashash. 2003. Evaluation of a permeable, 5 cm thick, polyethylene foam lagoon cover. Transactions of the ASAE 46:1421-1426.

Misselbrook, T. H., S. K. Brookman, K. A. Smith, T. Crumby, A. G. Williams, and D. F. McCrory. 2005. Crusting of stored dairy slurry to abate ammonia emissions: Pilot-scale studies. J. Environ. Qual. 34:411-419.

Roos, K.F., M.A. Moser, and A.G. Martin. 1999. Agstar charter farm program: experience with five floating lagoon covers. Fourth biomass conference of the Americas, Oakland, CA.

Rotz, C.A. 2004. Management to reduce nitrogen losses in animal production. J. Anim. Sci. 82:E119-E137.

Scotford, I.M., and A.G. Williams. 2001. Practicalities, costs and effectiveness of a floating plastic cover to reduce ammonia emissions from a pig slurry lagoon. Journal of agricultural engineering research 80:273-281.

Smith, K., T. Cumby, J. Lapworth, T. Misselbrook, and A. Williams. 2007. Natural crusting of slurry storage as an abatement measure for ammonia emissions on dairy farms. Biosys. Eng. 97:464-471.

Unknown. 2014. Reduction of algae and odours leads to increased capacity of lagoon. Environmental Science 7 Engineering Magazine, May/June 2014, pp22-23.

4.1.3 Floating Barriers

General Description

Floating barriers can be used to create preferential flow patterns in water bodies to facilitate the conditions needed for all phases of the nitrogen cycle: ammonification, nitrification, and denitrification. Floating barriers, also called baffles, are designed to increase the retention time and settling time in water treatment lagoons to reduce total

suspended solids and provide greater potential to enhance water quality treatment via aerobic and anaerobic processes that metabolize nutrients.

Figure 8. Floating Barriers. http://www.elastec.com/baffles/gallery

Issues Addressed

Floating barriers, or water control baffles, are used in industrial and municipal wastewater treatment process plants to control and improve water quality. Factors that influence the hydraulic performance in ponds are mainly related to shape, including baffles (Mangelson and Watters 1972). Generally, the design recommendations related to hydraulic performance emphasize the relationship between pond length, width, and depth, and also the location of inlets and outlets with different configurations. Length-to-width ratio is generally seen as the most important factor influencing hydraulic performance (Watters 1972; Persson 2000).

Regular treatment ponds and lagoons have large areas of poor water circulation outside the flow line between the inlet of untreated water and the outlet of the water body, which acts as ineffective dead space that does not treat water as successfully as areas with a higher retention time. Engineered baffles and barriers can redirect flow to utilize the entire area of the pond, avoiding “short circuiting” between inlet and outlet. The

data indicate that waste stabilization ponds should be constructed with large values of length to width ratio to give maximum detention time and the minimum value for dead space in order to achieve greater treatment of wastewater (Watters 1972). These devices are designed to result in better settling of TSS, improvement of BOD, increased retention time to remove nutrients, and improved mixing. By utilizing a greater amount of surface area with floating barriers, an older pond can maintain volume and work more efficiently.

Suitable baffle configurations may help form favorable flow fields and increase the efficiency of a primary sedimentation tank, by enhancing the uniform flow field in the settling zone when the baffle position provides a small volume of circulation regions (Shahrokhi et al. 2013). The floating and flexible design allows the floating barrier to adjust to changing fluid levels. Flow diversion curtains supported by floating baffles increase hydraulic retention time and can increase the effective treatment area of a lagoon from less than 20% to over 95% (Environetics, Inc. 2015). In a case study from a manufacturer in Illinois, a series of wastewater lagoons were experiencing short circuiting, as evidenced by ice formation around the edge of the lagoon and high levels of BOD and TSS that were close to exceeding typical standards 9 to 10 times per year (Environetics, Inc. 2015). The installation of a floating baffle system eliminated these problems and reduced their BOD and TSS excursions by 90% (Environetics, Inc. 2015). A wastewater lagoon is most likely to short circuit in winter, due to the disparity between the influent wastewater temperature and the ambient lagoon temperature. Studies have shown that long, evenly spaced baffles 70% of the width of the lagoon provide the greatest hydraulic efficiency, which gave superior performance over baffles that were 50% and 90% of the width of the pond (Watters et al. 1973). However, mathematical modeling may be required to determine the optimal size, number, and placement of baffles for a particular installation (Hill 2014). A later study showed that the channeling seen with 90% width baffles was dependent on baffle spacing as a result of the number of baffles in a pond (Shilton and Harrison 2003). Computer modeling showed that a single baffle does give an improvement over an un-baffled case, but increasing the number of baffles was far superior and a further, but smaller and unwarranted, improvement was achieved by increasing to four baffles (Shilton and Harrison 2002). A horizontal baffling scheme was more efficient than vertical baffling for a comparable number of baffles (Watters 1972). Furthermore, a pattern of standard, even spacing along the length of the pond proved most effective compared with alternative baffle layouts (Shilton and Harrison 2002). Feasible designs were previously obtained by trial and error to satisfy design constraints with little consideration of economy, but the design of a baffled channel has since been formulated as a geometric programming problem that yields a solution of the optimal design for the corresponding channel dimensions (Swamee 1996). A subsurface berm or an island placed in front of the inlet also improves the hydraulic performance concerning short-circuiting, effective volume, and amount of mixing (Persson 2000).

Figure 9. http://www.nelsonenvironmental.com/optaer/lagoon-baffles.asp

Design Variations

Barrier components are selected for their corrosive resistant properties, ability to withstand ultraviolet radiation, chemical resistance, and hydrodynamic conditions. Elastec American Marine Company manufactures floating baffle systems from fabrics including urethane and polyvinyl chlorides (PVC's), and are capable of building one piece baffle sections up to 2,500 feet (Elastec American Marine 2015). Other varieties of water control baffles are constructed from impermeable PVC (Granite Environmental 2015). Floating baffle curtains are hung from a top tension cable with flotation contained in the curtain fabric, and can also include specially sized and placed windows that control flow and circulation in wastewater treating ponds (Abasco 2015). A large (4,500 feet) curtain designed for a wastewater treatment plant on the Mississippi Gulf Coast contained flow-through ports positioned on the floating baffle curtain to allow the liquid to pass through while the surface debris could not pass (ETP 2015). Environetics recommends a serpentine arrangement of floating barriers using flow diversion curtains to maximize aeration and retention time while allowing contaminants and solids to settle, and additionally designs multicellular aerated lagoons that allow aerators to operate more efficiently (Environetics, Inc. 2015). The baffles can be used to isolate a quiescent settling zone to increase the BOD removal capacity of a lagoon system that is short on retention time (Nelson Environmental 2015). The ideal configuration for a pond designed on the basis of complete mix hydraulics is a circular square pond with cells configured with a length to width ratio of 3:1 or 4:1 (EPA 2011). The dimensions of the cells can be calculated using the following equation:

$$V = [LW + (L - 2sd)(W - 2sd) + 4(L - sd)(W - sd)]d/6$$

Where:

V = volume of pond or cell, meters^3
L = length of pond or cell at water surface, meter

W = width of pond or cell at water surface, meter
s = slope factor (e.g., with 3:1 slope, s = 3)
d = depth of pond, meter

(EPA 2011)

Alternately, Thackston and others. (1987) carried out an extensive study, investigating 12 ponds (shallow basins) with dye tracer to track flow. Most of the ponds were rectangular with length to width ratios varying from 0.7 to 4.1. Thackston and others (1987) recommended a length to width ratio of 10:1, while Reed and others (1995) recommend a ratio of 4:1 in a well-designed system.

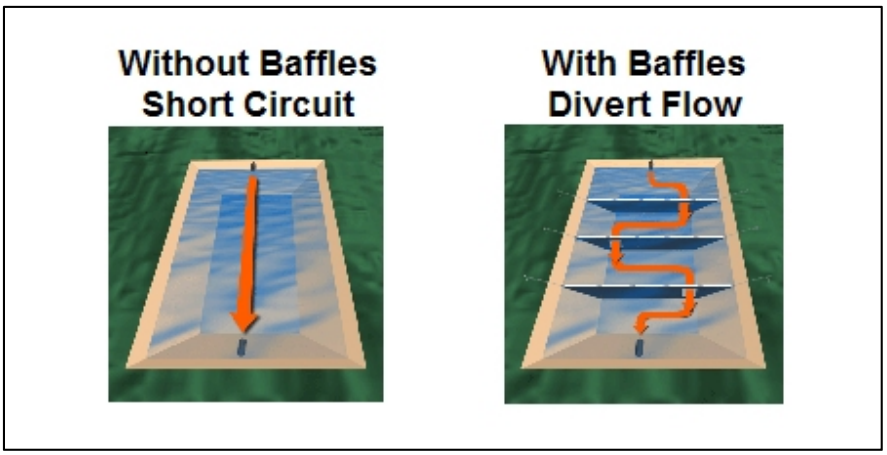

Figure 10. http://www.environeticsinc.com/FloatingBaffle.htm

A study using nonwoven geotextiles as a compact biomass host media in the form of suspended baffles organized in a sinuous channel with permeable boundaries was successful at treating wastewater from a combined sewer system (Korkut 2006). Filtering the TSS and micro-organisms formed a biomass floc in the interior of the baffles, which grew to emerge on the surface. Suspended and non-settleable colloidal solids in the influent wastewater were captured by both filtration and adsorption from the channel flow. This bench-scale setup, named the geotextile baffle contact system, consistently provided secondary treatment to influent concentrations up to 318 mg/L of TSS and 114 mg/L of BOD (Korkut 2006). Ammonia concentrations were reduced over 90%, and mineralization of the nitrate was also observed when the biofilm aged and thickened (Korkut 2006). Some of the influent TSS and sloughed biomass from the baffles settled to the bottom of the tank (Korkut 2006). A different study found that baffles significantly increase the amount of attached biomass, and at lower hydraulic residence times increase soluble organic carbon removal: with a 1.5-day hydraulic residence time, a pond without baffles removes 53% of the soluble organic carbon, while ponds with end-around, over-and-under, and longitudinal baffles remove 60%, 62%, and 70% soluble organic carbon, respectively (Reynolds et al. 1975).

Figure 11. http://www.environeticsinc.com/FloatingBaffle.htm

Operations and Maintenance Issues

Storms and high winds can move baffles around and in turn mooring anchors, which results in a mis-positioned baffle system (Chatoyer Environmental 2014). This may result in decreased effectiveness of the water flow and a return to short circuiting. Hence, the system should be regularly checked with routine maintenance to ensure its effectiveness, especially after large storm events. Typical inspection and maintenance activities should include: review and confirmation of the positioning of each baffle; review of the integrity of the fabric and connection points; replacement of worn or broken anchor lines; inspection and reporting on any sludge build-up which may be placing undue pressure on the system; review of hardware that is often placed under pressure; and review of possible fouling by birds or other animals (Chatoyer Environmental 2014).

Relative Capital Costs

A properly designed lagoon with floating barriers or baffles can be constructed at 40 to 50% less cost than activated sludge systems and use 50% less operating power, reducing overall energy costs while increasing aerator efficiency (The Environmental Company 2015). The floating baffles allow existing systems to be upgraded without the expense and difficulty of dewatering the lagoon to achieve the desired flow pattern. In a case study in Barboursville, West Virginia, the village increased capacity of the existing lagoon from 600,000 gallons per day (gdp) to 1,500,000 gdp with no disruption of treatment during construction and a total project cost of less than $250,000 with all work completed exclusively by Sanitary Board personnel (Environetics, Inc. 2015). Existing aerators were repositioned in 5 of the zones created by the new floating baffles, and a combination of gaseous chlorine and liquid sodium bromide were used to disinfect the lagoon's effluent (Environetics, Inc. 2015).

Constructability

Water baffles are designed to fit the dimensions and specifications of individual ponds, and are easily installed in existing lagoons and ponds without draining by using the baffle's built-in flotation as an aid to installation. Manufacturers require information on

basin dimensions, side slope, type and number of aerators, maximum flow, water temperature, and water pH (Abasco 2015). Baffle curtains can be designed to fit the specific contours of individual waterbodies, and are easily installed without costly equipment or labor. Curtains can be manufactured in multiple units and shipped to the job site for connection to a previously installed shore anchor, floating of the curtain into place (with closed cell flotation foam integrated into a sealed pocket formed by the baffle geomembrane), securing in the correct position to create optimal flow pattern, and attachment of ballast to the bottom of the curtain (ETP 2015). The installation of large floating baffles generally takes one day, although the anchor should be prepared several weeks in advance to ensure the concrete is fully hardened and the surrounded soil is compacted around it. The shore anchor, or mooring, is a permanent post made of metal and secured by concrete into the embankment, and can be further reinforced using either a screw anchor or a flat plate (Chatoyer Environmental 2014). Floating baffle curtains can be installed in wet or dry conditions, which increases the possibilities for adjustment and upgrading of old lagoon systems (Nelson Environmental 2015). The installation of a floating baffle within a lined pond will require a different form of anchoring so as to not damage the liner; a common approach is to deploy rubber tires filled with concrete (Chatoyer Environmental 2014).

Figure 12. http://www.nelsonenvironmental.com/optaer/lagoon-baffles.asp

<u>Recommendations</u>

Floating baffles/barriers are a relatively cost-effective method to ameliorate existing lagoon configurations with a more effective design that eliminates “short-circuiting” of wastewater flow. Many companies create custom-designed floating baffle and curtain setups that fit the specifications of individual lagoons and can be installed on-site by independent contractors or local maintenance staff. Attention should be paid to the dimensions of channels created by the baffles, with emphasis on the research and standards set concerning length to width ratios, notably a longitudinal orientation of baffles within the lagoon (Watters 1972) and 2 to 4 baffles, depending on the desired flow and mixing at the site (Shilton and Harrison 2003). Use of floating baffles/barriers has the potential to significantly increase the settling of suspended solids, improvement

of BOD, increased retention time to decrease nutrient concentrations, and improved mixing.

References

Abasco. 2015. Floating Baffle Curtains. Available at: http://www.abasco.com/baffles.html. Accessed April 15, 2015.

Chatoyer Environmental. 2014. Floating Baffle User Guide. Chatoyer Company Limited. Available at: http://www.chatoyerenvironmental.com.au/wp-content/uploads/2014/11/Floating-Baffle-User-Guide.pdf. Accessed April 15, 2015.

Elastec American Marine. 2015. Baffles. Available at: http://www.elastec.com/baffles/. Accessed April 15, 2015.

Engineered Textile Products. Floating Baffle Curtain. Available at: http://www.etpinfo.com/floating-baffle-curtain/Floating-Baffle-Curtain-Mississippi-Gulf-Coast.pdf. Accessed April 15, 2015.

Environetics, Inc. 2015. Floating Baffles. Available at: http://www.environeticsinc.com/FloatingBaffle.htm. Accessed April 15, 2015.

The Environmental Company: Wastewater Treatment Solutions. 2015. Baffle Walls/Curtain Walls. Available at: http://www.ebac2000.com/baffle.htm. Accessed April 15, 2015.

U.S. Environmental Protection Agency (EPA). 2011. Principles of Design and Operations of Wastewater Treatment Pond Systems for Plant Operators, Engineers, and Managers (PDF). (EPA/600/R-11/088) 10.76 MB, 457 pp.

Granite Environmental. 2015. Water Control Baffles. Available at: http://www.erosionpollution.com/water_baffle.html. Accessed April 15, 2015.

Hill, Patrick. 2014. How to prevent wastewater lagoon short circuit. Triplepoint's Lagoon Blog. Posted on December 22, 2014. Available at: http://www.triplepointwater.com/prevent-lagoon-short-circuit/#.VS_Y9OOsXA0 Accessed April 16, 2015.

Korkut, E., Martin, J., and Yaman, C. (2006). "Wastewater Treatment with Biomass Attached to Porous Geotextile Baffles." J. Environ. Eng., 132(2), 284–288.

Mangelson, K. 1971. Hydraulics of Waste Stabilization Ponds and its Influence on Treatment Efficiency. Doctorate Thesis; Department of Civil Engineering, Utah State University; Utah, USA.

Mangelson, K.A., and G.Z. Watters. 1972. Treatment efficiency of waste stabilization ponds. Journal of Sanitary Engineering Division, American Society of Civil Engineering, 98:2. Pp 407-425.

Nielson, Stephen B., E. Joe Middlebrooks, and Donald B Porcella. 1973. Effects of Baffles on the Performance of Anaerobic Waste Stabilization Ponds. Reports. Paper 172. http://digitalcommons.usu.edu/water_rep/172

Nelson Environmental, Inc. 2015. Lagoon Baffles. Available at: http://www.nelsonenvironmental.com/optaer/lagoon-baffles.asp. Accessed April 15, 2015.

Persson, J. 2000. The Hydraulic Performance of Ponds of Various Layouts. Urban Water 2:3, pp. 243-250.

Reed, S. H., R.W. Crites, and E.J. Middlebrooks. 1995. Natural systems for waste management and treatment (2nd ed.). New York: McGraw-Hill.

Reynolds, J.H., E.J. Middlebrooks, and S.B. Nielson. 1975. Biomass distribution and kinetics of baffled lagoons. Journal of the Environmental Engineering Division, 101:6, pp 1005-1024.

Shahrokihi, M., F. Rostami, A. Said, and S.R.S. Yazdi. 2013. Experimental investigation of the influence of baffle position on the flow field, sediment concentration, and efficiency of rectangular primary sedimentation tanks. Journal of Hydraulic Engineering, 139:1, pp 88-94.

Shilton, Andy, and Jill Harrison. 2003. Development of Guidelines for Improved Hydraulic Design of Waste Stabilization Ponds. Institute of Technology and Engineering, Massey University, New Zealand. Available at: http://www.bvsde.paho.org/bvsacd/agua2003/hidra.pdf. Accessed April 16, 2015.

Swamee, P.K. 1996. Design of flocculating baffled channel. Journal of Environmental Engineering, 122:11, pp 1046-1048.

Thackston, E.L., F.D. Shields, Jr., and P.R. Schroeder. 1987. Residence time distributions of shallow basins. Journal of Environmental Engineering, 113:6, 1319-1332.

Todd, J., E.J.G. Brown, and E. Wells. 2003. Ecological design applied. Ecological Engineering 20: 421-440. Available at: https://www.uvm.edu/giee/pubpdfs/Todd_2003_Ecological_Engineering.pdf

Watters, Gary Z. 1972. The Hydraulics of Waste Stabilization Ponds. Utah State University, Utah Water Research Laboratory; Reports: Paper 18.

Watters, G., K. Mangelson, and R. George. 1973. The Hydraulics of Water Stabilization Ponds. Research Report; Utah Water Research Laboratory, College of Engineering, Utah State University; Utah, USA.

4.2 Mechanical Techniques

Mechanical improvements to facultative lagoon systems include the addition of lagoon infrastructure and/or treatment processes in order to improve performance. These technologies typically include moderate to major design changes and variable capital costs depending on the selected technology. They also may include increased operation and maintenance requirements which should be considered.

4.2.1 Aeration

General Description

In facultative wastewater treatment lagoons, bacteria convert organic nitrogen to ammonia (or dissolved ammonium) under anoxic conditions in a biologic process called ammonification. Removing ammonia is integral to the treatment process, as excess ammonia in effluent streams places a significant BOD load on the receiving water. Ammonia removal in facultative lagoons is generally achieved through algal consumption of ammonium, volatilization of gaseous ammonia to the atmosphere, and biological nitrification.

Both volatilization and nitrification rely on mixing and oxygen in treatment lagoons. The rate at which ammonia volatilizes generally relies on the difference between ammonia concentration in the water at the lagoon surface and that in the atmosphere; therefore, mixing increases this rate by bringing water with higher ammonia concentrations to the surface and exposing it to the air.

In the biological nitrification process, bacteria in the facultative and aerobic zones convert ammonia to nitrite, then nitrite to nitrate. Since oxygen is consumed during these reactions, and since many lagoon systems are not well mixed, the concentration of DO in the water column can be the limiting factor in the nitrification process.

Aeration is a commonly used method of oxygen delivery and lagoon mixing in wastewater treatment lagoons. This review evaluates the design, operation and maintenance, and relative costs of several currently available aeration and oxygen delivery systems. Also included are evaluations of the constructability or ease of implementation of these systems, brief summaries of new and innovative aeration technologies, and recommendations that may be applicable to Montana communities that operate facultative wastewater treatment lagoons.

Issues Addressed

Poor mixing and low DO concentrations in wastewater treatment lagoons inhibit ammonium and ammonia removal, which can lead to several water quality problems in receiving waters, including:

- Ammonia-N, BOD, and TSS loading;
- Fish toxicity; and
- Noxious odors.

Aerators and other systems can significantly improve treatment efficiency in wastewater lagoons that currently rely exclusively on natural processes such as wind for oxygen and mixing.

Design Variations

The majority of wastewater treatment lagoons operating in Montana are facultative lagoons that rely on anaerobic biological decay of organic matter at the bottom of the lagoon. Therefore; oxygen should not be introduced to the anaerobic zone of these systems, and aeration should be limited to the lagoon surface in the primary facultative cell. Aeration and complete mixing is often implemented in secondary cells for this reason.

There are two main types of pond aerators: diffused aerators and mechanical aerators. Diffused aerators, or bottom aerators, transfer oxygen by injecting air bubbles. Oxygen is transferred across the surface of the bubbles and into the water.

Diffused aerators are made by several manufacturers in the United States, and several more design variations are available. Generally, diffused aerators fall into one of two categories, coarse bubble and fine bubble aerators. Both are typically constructed of stainless steel headers with air discharge ports. The diffusers themselves have no moving parts; a large compressor or regenerative blower supplies air to the diffusers, where it exits through the ports, is sheared into bubbles, and distributed into the liquid (RWLWater 2015).

A CYCLONE Coarse Bubble Diffuser, manufactured by RWLWater, is depicted below submerged in an aeration basin.

Figure 13. Cyclone Coarse Bubble Diffuser

As the name implies, fine bubble diffusers operate under the same principles as coarse bubble diffusers but inject smaller bubbles into the water. Fine diffusers have higher oxygen transfer efficiency (OTE) than coarse systems, since the same volume of injected air produces more bubble surface area using smaller vs. larger bubbles. Since, theoretically, fine bubble systems can achieve the same oxygen transfer as a coarse system with less air injected, the literature unanimously describes fine bubble diffusers as being more energy efficient than coarse bubble diffusers.

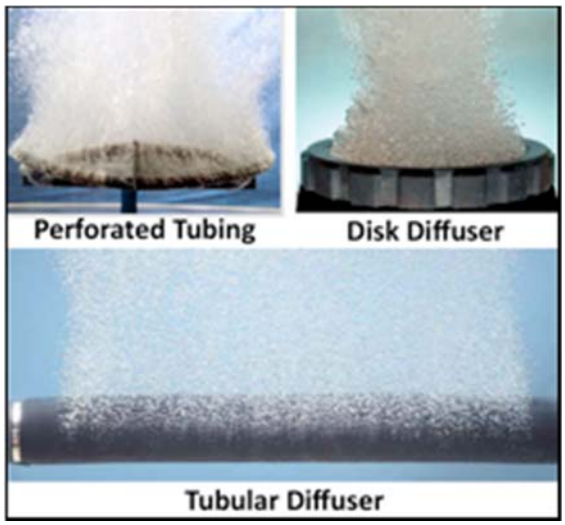

Figure 14. Three main types of fine bubble diffusers (Triple Point).

Disadvantages of fine bubble diffusers include susceptibility to fouling (EPA 1999), biological or chemical degradation, uneven air distribution throughout the diffuser, and most importantly, difficulty of installation and maintenance. Fine bubble diffuser systems are either fixed to floating laterals or fixed laterals at the lagoon bottom. In the

first case, the system is installed and maintained by boat. In the second case the lagoon must be drained to install and maintain the diffuser.

Mechanical aerators (or surface aerators) operate on the water surface and function by agitating the water to create small water droplets. Oxygen is transferred across the surface of the droplet from the surrounding air. Although much of the literature discusses the many advantages of diffused aerators over surface aerators (for example: oxygen delivery performance, energy efficiency, and less complex maintenance), surface aerators may applicable for use in existing facultative lagoons since they can be deployed to aerate the upper aerobic zone without introducing oxygen to the deeper anaerobic zone. It is important to note that oxygen transfer is much less efficient at shallower depths.

Surface aerators agitate the water using blades, propellers, or brushes, and can be fixed systems or floating aerators. RWLWater manufactures several surface aerators, which include the following:

- The TORNADO™ Surface Aspirating Aerator consists of a motor, a draft tube, and a propeller. When the propeller rotates, it pushes the water forward, creating a low-pressure zone at the propeller hub. This draws air through the air inlet hole, down through the draft tube, until it reaches the propeller. The rotating action of the propeller mixes the air stream with the water. A blower can be added to inject additional air and increase oxygen delivery.
- The TWISTER™ Low-Speed Surface Aerator consists of a motor mounted to a gear reducer and an extended shaft. A rotor is attached to the bottom of the extended shaft and partially submerged in the wastewater. This aerator turns at a slow speed (typically 40 to 100 rotations per minute (rpm)) and fins on the rotor then lift water into the air in a fine spray.
- The TYPHOON™ High Speed Floating Aerator supports a motor vertically and above the water surface. The motor drives an extended shaft with a submerged propeller which draws water up through the aerator intake cone and projects it outward, above the water line in a low trajectory spray.
- The MONSOON™ Horizontal Surface Aerator utilizes rotating blades that turn at 40 to 80 rpm and scoop water into the air. This scooping and mixing action breaks the water droplets into a mist.

New/Innovative Approaches

Philadelphia Mixing Solutions manufactures several surface aerators including: directional mixing aerators, brush rotors, and wave generators; which increase the effective surface area of the water, thus increasing oxygen diffusion from the air. The OLOID wave generating aerator is shown below.

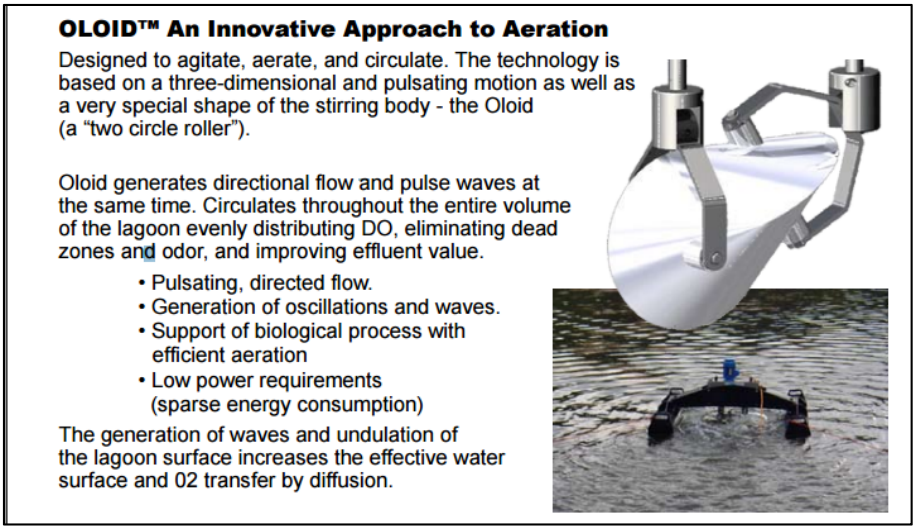

Figure 15. OLOID aerator by Philadelphia Mixing Solutions

Operations and Maintenance Issues

While diffusion aerators are becoming more popular for use in wastewater lagoons, these systems are secured using either fixed laterals at the lagoon bottom or floating laterals. If fixed laterals are used, the lagoon would need to be drained to install and maintain the system (Hill 2013). If floating laterals are chosen, installation and all future maintenance would take place by boat.

Mechanical aerators function using continuously revolving motors and blades or jets at the lagoon surface. They are easily installed by guiding the aerator into position and anchoring it. Operation and maintenance is more intensive; however, due to more frequent preventative maintenance and increased potential for failure due to continuously moving parts, and the fact that the systems are typically serviced from a boat (Hill 2013). Splashing and mist can freeze in the winter and cause motor failure. The climate in many parts of Montana may preclude the use of many surface aerators without additional protection from freezing.

Relative Costs

The capital costs associated with diffusion aerators can be assumed to be higher than those of mechanical aeration systems. Both systems employ the use of expensive motors, blowers, or compressors; however, installing a diffusion system involves either draining and desludging the lagoon followed by installation by a construction crew, or crews installing the system by boat at the surface.

When comparing operating costs, fine bubble diffusers are generally accepted as the most efficient system due to fewer equipment failures, less complex maintenance, and energy efficiency as shown in the figure below (Fisher 2011). However, to realize the potential savings brought by the increased efficiency of fine bubble diffusers, operators may need to install expensive automatic control systems (Bolles 2010). Common control components include DO probes, primary logic controllers, and variable speed drives on air compressors or blowers.

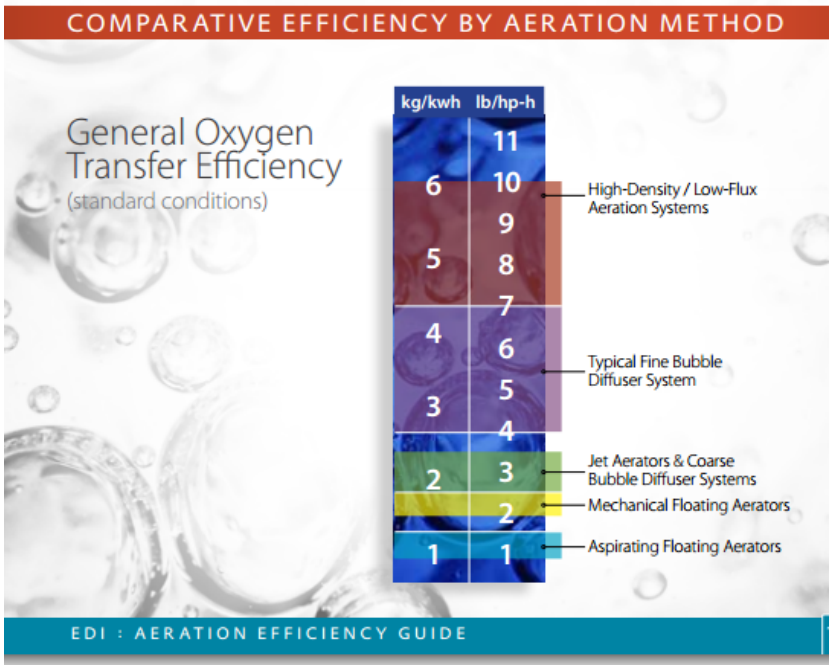

Figure 16. Aeration Efficiency Comparison (EDI 2015).

It should be noted that the literature typically assumes that the same level of oxygen delivery would be desired from a mechanical system as from a diffuser system. However, in reality, operators may be willing to sacrifice some efficiency in energy consumption and oxygen transfer if a relatively small mechanical system would reduce overall operating cost. Further, in the case of facultative lagoon cells, oxygen delivered to the bottom of the lagoon would inhibit anaerobic biodegradation and is not desired.

Constructability

Aeration and other oxygen delivery systems discussed in this review all appear to be relatively uncomplicated to install in most wastewater lagoons. However, the need to either drain a lagoon or use boats to install diffusion aerators add considerable difficulty to this option.

Recommendations

The aeration technologies evaluated for this review all increase DO and, to some extent, provide mixing in lagoons. They are all relatively simple to install and are scalable to suit different populations and lagoon sizes, and should not require additional land area to implement. However, all involve the addition of permanent infrastructure with associated operation, maintenance, and electrical power costs. The most desirable specific technology depends heavily on the current lagoon design, retrofit capabilities, and treatment objectives in each lagoon.

For facultative systems, any of the available floating mechanical systems may be desirable; they are simple to install, provide aeration and mixing to the aerobic zone, and can be scaled to fit different populations and lagoon sizes by varying the number of aeration units deployed. The more aggressive surface aerators may not be well suited

to the climate in much of Montana due to their increased potential to fail from freezing, but several "low splash" or "no splash" models are available. All mechanical systems have considerably higher operating costs per unit of oxygen delivered than the more efficient diffusive systems. If ammonification is severe and significant increases in DO concentration are required, retrofitting a lagoon to include an aerated secondary cell that utilizes diffusion aerators may be the preferred option. Montana Circular DEQ-2 requires that aerated lagoons be at least 10 feet deep. The addition of aeration systems in shallow facultative lagoons may impact existing sludge layers, potentially re-suspending solids and releasing nutrients.

References:

Bolles. 2010. Modeling Wastewater Aeration Systems to Discover Energy Savings Opportunities, Steven A. Bolles, Process Energy Services, LLC.

U.S. Environmental Protection Agency (EPA). 1999. Wastewater Technology Fact Sheet, Fine Bubble Diffusers, United States Environmental Protection Agency, September 1999.

Fisher. 2011. Aeration Advantage, Zeev Fisher, Water & Wastes Digest, October 2011.

Hill. 2013. Fine Bubble Aeration in Wastewater Lagoons – Pros and Cons. Patrick Hill, Triplepoint Water Technologies, July 2013.

RLW Water. 2015. Available at: http://www.rwlwater.com

4.2.2 Fixed-film Bioreactors

General Description

Fixed film bioreactors, often referred to as attached growth reactors, are widely used in the wastewater treatment industry. Fixed film reactors use some type of media, usually a porous, inert substance, which provides a structure to support the bio-film that digests the waste material in the wastewater. Designs for fixed film systems vary widely, but fall into two basic categories: 1) A system where the media is moved relative to the wastewater (fixed media); and 2) A system that incorporates stationary media, and wastewater flows over the media (suspended media). The support media can be made of any suitable porous material, such as: formed plastic, rocks, sand, and numerous other manmade and natural substances. Attached growth reactors can be either aerated (aerobic) or non-aerated (anaerobic) depending on the type of treatment required. Some basic types of fixed film reactors relevant to facultative lagoons include: trickling filters, packed bed reactors, fluidized bed reactors, moving bed reactors, and submerged aerated reactors. Each is described in more detail below.

According to Jenkins and Sanders (2012), Fixed-film reactors have some advantages to suspended growth treatment technologies, including:

- Reduced operating and energy costs;
- Smaller reactor volume;
- Minimized need for settling capacity;
- Operational simplicity; and

- Reduced sludge.

Issues Addressed

Fixed film biological reactors are capable of addressing multiple issues associated with facultative lagoons. This technology is widely used in the wastewater industry to treat ammonia, total nitrogen, BOD, and TSS. Attached growth reactors greatly enhance the nitrification and denitrification reactions, which also reduce BOD.

Design Variations

A single biofilm may have aerobic, anoxic, and anaerobic processes occurring, and the substrate that becomes limiting will change through the depth of the biofilm. Because of this, fixed-film processes are quite complex to model. A variety of media systems have been used and several types have become standard in the industry. In general, the media types may be differentiated as either fixed or free floating.

Fixed Media Systems. In fixed media systems, the media is stationary and the wastewater is moved around the media. A fixed media filter mimics the treatment method used in a healthy natural ecosystem in which microorganisms produce a slick coating on rocks and pebbles. This coating of microorganisms is able to trap and consume contaminants in the water. There are several types of fixed media systems used in wastewater treatment including: trickling filters, rotating biological contactors, packed bed reactors, and submerged attached growth reactors.

Trickling Filter. Trickling filters are very efficient at removing BOD and ammonia from wastewater capable complete BOD removal and nearly complete nitrification. A trickling filter is a three-phase system with fixed biofilm carriers. Wastewater enters the bioreactor through a distribution system, trickles downward over the biofilm surface, and air moves upward or downward in the third phase. Trickling filter components typically include a distribution system, containment structure, rock or plastic media, underdrain, and ventilation system. Wastewater treatment using the trickling filter results in a net production of TSS. Therefore, liquid-solids separation is required, and is typically achieved with circular or rectangular secondary clarifiers. The trickling filter process typically includes an influent pump station, trickling filter, trickling filter recirculation pump station, and liquid-solids separation unit.

Ideal trickling filter biofilm carriers, or media, provide a high specific surface area, low cost, high durability, and high enough porosity to avoid clogging and promote ventilation (Tchobanoglous et al. 2003). Trickling filter biofilm carriers include: rock, random (synthetic), vertical-flow (synthetic), and cross-flow (synthetic). Both vertical-flow and cross-flow media are constructed with smooth and/or corrugated plastic sheets. Another commercially available synthetic media, although not commonly used, are vertically hanging plastic strips. Horizontal redwood or treated wooden slats have also been used, but are typically no longer considered because of their high cost or limited supply. Modules of plastic sheets (i.e., self-supporting vertical-flow or cross-flow modules) are used almost exclusively for new and improved trickling filters. However, several trickling filters with rock media exist and are capable of meeting treatment objectives when properly designed and operated. The higher specific surface area and

void space in modular synthetic media allow for higher hydraulic loading, enhanced oxygen transfer, and biofilm thickness control in comparison to rock media.

Nitrifying trickling filters are a reliable and cost-effective means for ammonia conversion. The following design practices have been demonstrated in full-scale application: (1) Use medium-density cross-flow media to optimize hydraulic distribution and oxygenation; (2) Use mechanical ventilation; (3) Periodically alternate the lead nitrifying trickling filter (NTF) to avoid patchy biofilm development in the lower reaches of the second-stage unit; (4) The influent should be secondary effluent to minimize bacterial competition for substrates inside the biofilm; (5) Maximize wetting efficiency to avoid the formation of dry spots; (6) Dose the NTF at a rate that will minimize the accumulation of macro fauna; and (7) Equalize ammonia laden supernatant from solids processing operations to even out diurnal load variability (Parker et al. 1995; 1997).

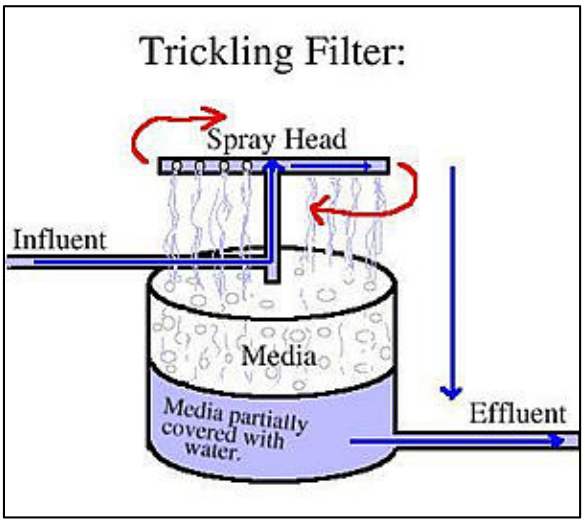

Figure 17. Trickling Filter

Rotating Biological Contactors (RBC). A Rotating Biological Contactor (RBC) is a biological treatment process used in the treatment of wastewater following primary treatment. The primary treatment process removes the grit and other solids through a screening process followed by a period of settlement. The RBC process involves allowing the wastewater to come in contact with a biological medium in order to remove pollutants in the wastewater before discharge of the treated wastewater to the environment, usually a body of water (river, lake or ocean). A rotating biological contactor is a type of secondary treatment process. It consists of a series of closely spaced, parallel discs mounted on a rotating shaft which is supported just above the surface of the wastewater. Microorganisms grow on the surface of the discs where biological degradation of the wastewater pollutants takes place. A RBC is an attached-growth biological process that consists of one or more basins in which large, closely-spaced circular disks mounted on horizontal shafts rotate slowly through wastewater. The disks, which are made of high-density polystyrene or PVC, are partially submerged in the wastewater, so that a bacterial slime layer forms on their wetted surfaces. As the disks rotate, the bacteria are exposed alternately to wastewater, from which they absorb organic matter, and air, from which they absorb oxygen. The rotary movement also allows excess bacteria to be removed from the surfaces of the disks and maintains a

suspension of sloughed biological solids. A final clarifier is needed to remove sloughed solids. Organic matter is degraded by means of mechanisms similar to those operating in the trickling filters process. Partially submerged RBCs are used for carbonaceous BOD removal, combined carbon oxidation and nitrification and nitrification of secondary effluents. Completely submerged RBCs are used for denitrification.

Figure 18. Image Source: OVIVO® RBC

Packed Bed (Fixed Bed) Reactors. Packed bed reactors, often called fixed bed reactors, are widely used in numerous industries, including wastewater treatment. Fixed-film anaerobic reactors have been widely used for the treatment of high strength wastewaters. In fixed-film anaerobic reactors, large amount of biomass remains in the filter to secure solids retention despite a short hydraulic retention times. These reactors have several advantages over aerobic and anaerobic reactors such as higher organic loadings, lower hydraulic retention times and smaller reactor volumes. Lower sludge and suspended solid quantities can also be achieved in these reactors (Kocadagistan et al. 2005). A combined anaerobic–aerobic fixed-film bioreactor has been tested for organic matter removal for 133 days and 92% removal efficiencies of organic matter removal achieved at an average of 0.39 kilograms COD/cubic meter per day organic loading rates (Del Pozo and Diez 2003).

There are numerous configurations for packed bed reactors and the desired results often dictate the design. In the wastewater treatment industry, the most common configuration is flooded, up-flow, anaerobic reactors.

Submerged Aerated Reactors (SARs). Within the wastewater industry submerged aerated reactors are often viewed as the simplest and most cost effective method of commercial and residential wastewater treatment, particularly for small to medium sized treatment plants where available land is limited, and where full time operational manning would be uneconomical. An adequately designed and built submerged aerated reactor plant has no moving parts within its main process zones; all serviceable items are positioned for easy access without disrupting the ongoing sewage treatment. The main advantages of SARs include:

- Predictable performance,
- Lower operation and maintenance costs,

- No interruption to process,
- Better whole life cost, and
- Easy installation.

SARs are used to reduce the organic loading of residential and commercial sewage/wastewater, and in doing so will reduce the BOD and a significant quantity of TSS; which if otherwise untreated, would contaminate river and sea outfalls. In other words, it is used to substantially improve effluent discharge quality. The usual effluent quality is typically better than 30 mg/L BOD; 100 mg/L TSS; and 10 mg/L ammonium.

As with traditional sewage treatment, SARs use three stages of treating wastewater:

- Primary Settlement,
- Secondary Treatment, and
- Final Settlement/Clarification.

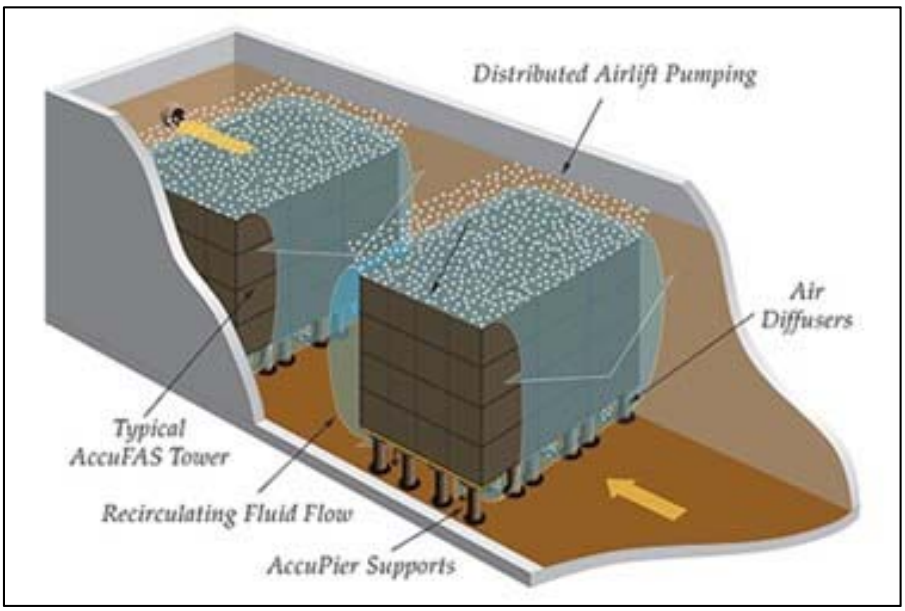

Figure 19. Source: Brentwood Industries, Inc. (2015).

Suspended (Free Floating) Media Systems. In suspended media systems, the media moves either with the movement of the fluid or with the addition of air to the wastewater stream. The microorganisms attach themselves to the media and treat the water, similarly to the fixed media systems. Suspended media systems are less prone to plugging and often have higher performance that fixed media systems. There are several major types of suspended media systems used in wastewater treatment, including: trickling filters, RBCs, packed bed reactors, and submerged attached growth reactors.

Moving Bed Bio-Reactors (MBRR). An MBBR is an improved version of fixed film bio reactor. The main difference is that the MBBR carrier media is free floating in the wastewater. MBBR technology employs thousands of polyethylene biofilm carriers operating in mixed motion within an aerated wastewater treatment basin. MBBR

processes are an excellent solution for common wastewater applications including: BOD reduction, nitrification, denitrification, and TSS.

An MBBR typically consists of a tank with submerged but floating plastic (usually HDPE, polyethylene, or polypropylene) media. The large surface area of the plastics provide abundant surface for bacterial growth. Biomass grows on the surface as a thin film whose thickness usually varies between 50 to 300 microns. Each individual bio carrier increases productivity through providing protected surface area to support the growth of heterotrophic and autotrophic bacteria within its cells. It is this high-density population of bacteria that achieves high-rate biodegradation within the system, while also offering process reliability and ease of operation. Medium or coarse bubble diffusers uniformly placed at the bottom of the reactor maintains a DO concentration of more than 2.5 to 3 mg/L for BOD removal. Still higher DO concentrations are maintained for nitrification. A clarifier is placed downstream of the MBBR tank to separate the biomass and the solids from the wastewater. No sludge recycle is required for this process.

Figure 20. Moving Bed Bioreactor

Wastewater enters the MBBR where the biomass attached to the surface of the media degrades organic matter resulting in BOD removal and/or nitrification depending on the type and characteristic of the wastewater. Organic carbon is converted to carbon dioxide and leaves the system while the ammonia and nitrogen in the organics are converted to nitrates through nitrification process. Oxygen required for the process is provided through the diffusers installed at the bottom of the reactor. The treated wastewater then flows through the screens to the downstream clarifier where the biomass and solids are separated from the wastewater.

For denitrification, anoxic MBBR tanks are used. The anoxic tank is similar to aerobic MBBR tank described above except that no oxygen is supplied. The tank contains no

diffusers and the media is kept in suspension in the reactor through submersible mixers. Other feature such as the screen is the same as aerobic reactor although the design may be slightly different.

Fluidized Bed Reactors. A fluidized bed reactor is a combination of the two most common: packed-bed and stirred tank continuous flow reactors. It is very important to many industries because of its excellent heat and mass transfer characteristics. In a fluidized bed reactor, the substrate is passed upward through the immobilized media bed at a high enough velocity to lift the particles. However, the velocity must not be so high that the support media are swept away from the reactor entirely.

Mendonca and others (2004) describes the performance, sludge production and biofilm characteristics of a full scale fluidized bed anaerobic reactor (32 cubic meters) for domestic wastewater treatment. The reactor was operated with a 10.5 m^3/hr up flow velocity, 3.2 hour hydraulic retention time, and recirculation ratio of 0.85. It presented removal efficiencies of 71% (+/-8%) of COD and 77% (+/-14%) of TSS. During the apparent steady-state period, specific sludge production and sludge age in the reactor were 0.116 (+/-0.033) kg VSS./Kg COD and (12+/-5)d, respectively. Biofilm formed in the reactor presented two different patterns: one of them at the beginning of the colonization and the other of mature biofilm. These different colonization patterns are due to bed stratification in the reactor, caused by the difference in local-energy dissipation rates along the reactor's height, and density, shape, etc. of the bio particles. The biofilm population is formed mainly of syntrophic consortia among sulfate reducing bacteria, methanogenic archaea such as *Methanobacterium* and *Methanosaeta*-like cells.

An (2001) used bench-scale fluidized-bed reactors with the immobilized activated sludge process to study the treatment of domestic wastewater. Different intermittent aeration patterns were investigated in order to improve the total nitrogen removal. The best total nitrogen removal at 74.4% was achieved at a hydraulic retention time (HRT) of 6 hours (with corresponding BOD loading rate of 0.766 kg/m3/d) and an aeration-pattern of 1, 3 (the hours of aeration and non-aeration time cycle). The removal efficiencies for organic carbon, ammonium, total Kjeldahl nitrogen, and TSS were not affected and remained at more than 90%. Simultaneous organic carbon and nitrogen removal was accomplished in a single immobilized sludge reactor. The impact of various influent feed patterns on the treatment was examined. The continuous feed pattern was recommended as it ensured good total nitrogen removal without any adverse impacts on the removal of organic carbon, ammonium, total Kjeldahl nitrogen, and TSS. The immobilized sludge beads exhibited satisfactory mechanical stability without apparent breakage over the 180-day experiment period.

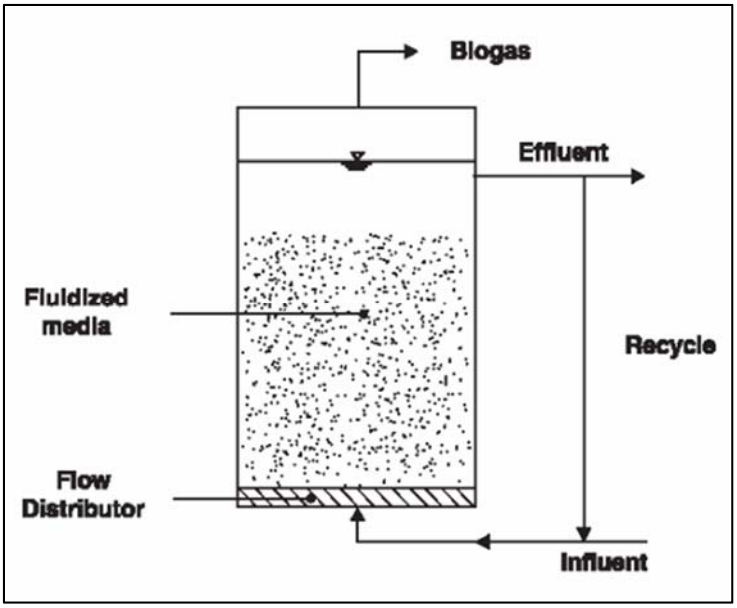

Figure 21. Fluidized Bed Reactor

Operations and Maintenance Issues

Very few operations and maintenance issues are associated with fixed film reactors. The main design considerations include: potential clogging of the media system as a result of inadequate screening; excessive growth, which could plug the media system or cause free-floating media to sink; and inadequate mixing or short-circuiting, resulting in inefficient use of the media. Many of the issues are easily overcome by developing and following a site-specific operations and maintenance plan.

Relative Capital Costs

The relative capital costs of upgrading facultative lagoons with fixed film technologies to achieve current water quality standards is in the low medium range compared to conventional mechanical wastewater treatment plants. Depending on the wastewater flow, adding an integrated fixed film system to an existing lagoon would be between 50% and 70% of the cost of a complete mechanical plant and could achieve approximately the same water treatment.

Constructability

The constructability of adding a fixed film system to existing lagoons has been demonstrated at numerous sites throughout the country. However, with the variation in size and design of the facultative lagoons in Montana, the actual constructability of adding a fixed film system would be evaluated on a case-by-case basis. Many of the technologies and currently available systems are designed to use the existing footprint of the lagoon and can typically be constructed without significant modifications. Constructability is also dependent on the required treatment levels that need to be achieved.

Innovative Approaches/Existing Technologies

The following are representatives of the commercially available systems that have the potential to work in Montana. The information provided comes directly from product literature or company websites, unless otherwise stated.

Wastewater Compliance Systems (bio-domes or Poo-Gloos). Wastewater Compliance System's patented bio-domes are intended for use as a means of increasing the naturally occurring biological activity in a water body. Bio-domes enhance the naturally occurring biology by providing 2800 square feet of surface area per unit, a gas delivery mechanism capable of providing air or more specific gasses as necessary, an environment protected from UV light, and significant mixing action to ensure nutrient availability to the biology as it grows.

When used in a wastewater lagoon, the enhanced biology is capable of accelerating the nitrification of ammonium and the subsequent denitrification of nitrates and nitrites. Additionally, the gas delivery/aeration design helps reduce BOD levels, and as organic material is consumed by the bacteria overall TSS levels are reduced. The airlift action created within the bio-domes is the mechanism behind the mixing action and helps delay the buildup of sludge on the bottom of the lagoon.

The USDA 2010 grant summary describes a recent advancement in enhanced biological uptake (poo-gloos™). Standard facultative lagoons

> *" are generally ineffective at nutrient removal, specifically nitrogen and phosphorous compounds dissolved in the discharge water. The biological removal of nitrogen compounds takes place in two stages when naturally occurring nitrifying, then denitrifying bacteria consume them. Biological removal of dissolved phosphorous compounds occurs through uptake by a class of bacteria known as polyphosphate accumulating organisms (PAOs). In open lagoon systems, these nutrient consuming bacteria are generally out-competed by the more robust bacteria that consume the carbonaceous BOD, and do not proliferate. To promote the proliferation of nitrifying and denitrifying bacteria and PAOs in shallow lagoon systems, specially designed aerated dome structures (Poo-Gloos) have been developed. These dome structures are nested hemispheres made out of ABS black plastic. Fine-bubble diffuser hose is attached in the gap between the inner and outer annulus around the bottom of each dome. Nested dome structures are mounted on a sturdy base and rest on the bottom of a lagoon so they are fully submerged. The space between the dome shells is filled with a high surface area to volume packing material made out of polypropylene plastic. Low-pressure air is supplied to the diffusers and the released bubbles travel up the inside of each dome. The dome structures retard the upward movement of the aeration bubbles, forcing the bubbles against naturally occurring bio-film colonizing the surfaces, promoting oxygen transfer. The rising air bubbles also drag the wastewater up from near the bottom of the lagoon, along through the insides of the dome structures, and out the top. This promotes micro-mixing of nutrients into the bio-film. By providing surface area, oxygen, mixing, and blocking of sunlight, these dome structures greatly enhance the growth and metabolism of nitrifying bacteria.*

In addition, denitrifying bacteria proliferate in the deeper portions of the inside bio-film surfaces as well as the non-aerated backside surfaces of the domes and during times the aeration bubbles are cycled off. The performance of PAOs is also enhanced by aeration cycling. This cycling should provide the necessary aerobic/anoxic/ anaerobic phases for these bacteria."

Figure 22. Courtesy of Wastewater Compliance Systems (2015) (Poo-Gloos).

Aeration Industries (Aire-O2 Bio-ffilm®). Aeration Industries' Aire-O2 Bio-ffilm® fixed film media system offers a cost-effective wastewater treatment facility upgrade that increases plant performance and capacity and facilitates year-round nitrification. The Bio-ffilm system is a submerged attached-growth media process using proprietary geotextile technology with a high effective surface area.

Our engineers have integrated the media into a treatment system using Aire-O2 Triton® aerators to establish and maintain biological growth on the media. The aerators are sized and oriented relative to the media arrangement to provide the proper amounts of mixing and aeration for solids suspension, nutrient treatment, and oxygen diffusion. The aerators also maximize treatment kinetics by maintaining higher basin temperatures in cold weather. The system is ideal for municipal wastewater treatment facilities needing advanced treatment upgrades to meet more stringent EPA requirements or capacity increases.

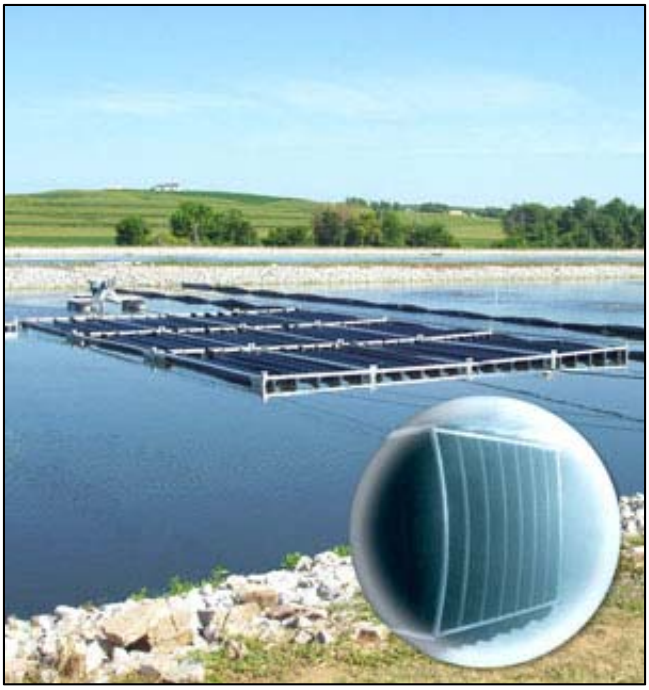

Figure 23. Courtesy of Aeration Industries (2015) (Aire-O2 Bio-ffilm®).

Lemna Technologies, Inc. (Lemna Polishing Reactor). The Lemna Polishing Reactor (LPR) provides effective supplemental BOD removal and ammonia polishing for pond-based municipal and industrial wastewater treatment facilities. The LPR is an up flow, attached-growth reactor. Using well-proven attached-growth media technology, the LPR enhances the growth of nitrification bacteria that biologically oxidize ammonia in an aerobic environment.

The LPR can be used as a stand-alone package treatment system for low loaded/small flow applications, or as an add-on to existing treatment processes. The LPR has proven to be an economical and reliable solution for achieving stringent effluent limits. Whether added as a final polishing step for supplemental BOD removal, or for ammonia removal, the LPR is often the most cost-effective tertiary treatment process available.

- **Easy to operate.** The Lemna Polishing Reactor was designed for simple operation, requiring virtually no operational interface. A small positive displacement blower is all that is needed to deliver aeration and mixing to the reactor.
- **Low energy consumption.** With no internal moving parts and efficiently aerated fixed film attached growth media, the energy cost to operate the LPR is minimal.
- **Flexible design**. The reactor can be economically sized to handle flows from 20,000 gpd to 2,000,000 gpd. The process allows for a wide range of influent BOD and ammonia loadings.

Figure 24. Courtesy of Lemna Technologies, Inc. (2015) (Lemna Polishing Reactor).

Nelson Environmental (SAGR Process). The (Submerged Attached Growth Reactor) SAGR® is an aerated gravel bed reactor, with a horizontal-flow hydraulic profile. The module provides year-round nitrification well beyond most total ammonia permit requirements. The SAGR® is designed to follow lagoons or other secondary treatment processes to provide nitrification in cold water temperatures. The SAGR® reliably achieves effluent total ammonia levels below 1.5 mg/L at water temperatures below 0.5°C or 32.9°F, and below 1.0 mg/L at warmer water temperatures. Nitrification (ammonia conversion to nitrates) in the OPTAER® system is performed by the SAGR®. The SAGR® is ideally suited for post lagoon nitrification in severe cold to moderate climates where water temperatures can be below 0.5°C for extended periods of time. The SAGR® can be combined with aerated or facultative lagoons to meet effluent ammonia requirements in systems that operate as continuous discharge or controlled discharge. The SAGR® was developed primarily to provide post-lagoon ammonia removal without abandoning existing lagoon treatment infrastructure. The performance parameters and sizing for the process are based on extensive testing performed on the post-lagoon SAGR® in Lloydminster and a demonstration SAGR® that was located in Steinbach, Manitoba. The SAGR® can be constructed within existing lagoon basins, or as an additional treatment module following a lagoon.

Figure 25. Courtesy of Nelson Environmental, Inc. (2015) (SAGR®).

EDI (IDEAL™ Solution). The Intermittently Decanted Extended Aeration Lagoon, or IDEAL, consists of an Environmental Dynamics International (EDI) floating lateral aeration system with Magnum fine bubble diffusers, two chains of BioReef BioCurtain, a static decanter with flow control valves, an overflow pipe with Storm Mode™, process controls, and a blower package. The simple operation of the IDEAL provides easy, cost-effective operation while the sophistication of the design provides high levels of BOD, TSS and ammonia removal. The process requires no active sludge management and extremely high flow events present no operational problem or concern from washout of the reactor. The IDEAL also provides up to 66% total nitrogen removal as part of the basic package, which means the owner is already prepared for future increases in nitrate or nitrogen requirements. The unique ability of the IDEAL to provide front-of-plant treatment provides several benefits over other lagoon-based technologies. First, the warmest water in the winter is found in the first cell. By performing treatment in the first cell the need for thermal covers is reduced. Second, by removing ammonia at the front of the plant the system can utilize the influent carbon for denitrification, which provides oxygen and alkalinity recovery while helping to meet future regulations without having to incorporate expensive future components. Lastly, because the sludge is retained in the first cell there is no need to operate and maintain sludge return pumps.

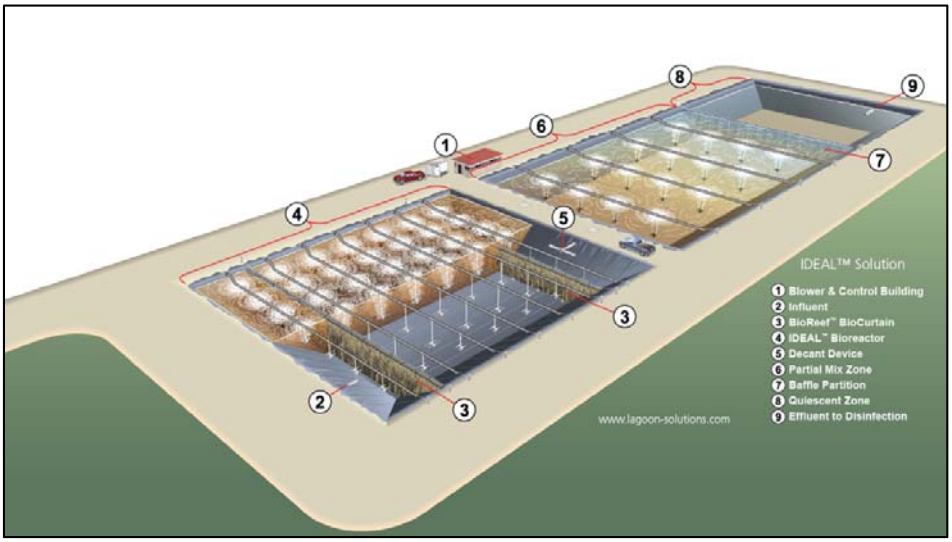

Figure 26. Courtesy of Environmental Dynamics International (EDI) (2015) (IDEAL™ Solution).

FBC Technologies, Inc. (Bio²Bloc®). FBC Technologies' Bio²Bloc was developed for aerobic wastewater retention basins, tanks, or lagoons that can benefit from fixed-film treatment.

The Bio²Bloc system has been in service since 1999 in municipal and industrial wastewater treatment plants. The system has performed successfully in removing BOD and ammonia – where alternatives would have been too expensive or have taken too long to implement. The Bio²Bloc provides a rich and stable environment to bolster your existing microorganisms. Unlike an RBC, the aerated media beds are in constant contact with the water and are completely aerated by a highly efficient aeration system.

Each Bio²Bloc is filled with thousands of pieces of bio-media where, during normal operation, a healthy biomass is formed by a supply of air and water produced by the fine bubble membrane diffusers located on the bottom of each bio-chamber. As with any fixed-film biological contactor, the units will tend to become overgrown with organisms after a period of time. When this happens, it's time for back-flushing. The patented Bio²Bloc has a separate coarse-bubble system, valved from the shore, to slough the media bed and clear the chamber for renewed growth.

Bio²Blocs are designed to be deployed individually or in series to remove pollutants. They diffuse a high level of DO into surrounding wastewater through the media bed. The units are built to specification and are easily installed using a light-duty boom truck and floated into place.

Figure 27. Courtesy of FBC Technologies, Inc. (2015) (Bio²Bloc®).

Meridian Aquatic Technology (Aquamats®). AquaMats® biofiltration curtains provide surface area to increase biomass concentrations for enhanced BOD, ammonia, and TSS reduction. AquaMats® are high surface area media that are supported on the water surface with ballast tubes. Each AquaMat® provides surfaces designed for biofiltration that extend into the water column. The units readily form into an array that extends across the width of the cell, parallel to and straddling the aeration system. This formation allows all water, which is being vertically mixed by the ADS aeration system to pass by the AquaMats® resulting in maximum wastewater contact with the biological communities growing on the AquaMats®. AquaMats® also function as hydraulic barriers that substantially increase the settlement of suspended solids for bacterial fixation in benthic sludge.

The principal of operation is to generate concentrated biomass (aerobic or anaerobic) and accelerated biological treatment in a dynamic process similar to the activities in attached growth systems such as trickling filters or RBCs.

The key to the success of AquaMats® is that they are specifically engineered to provide an ideal location for the beneficial microbial life forms that naturally clean and purify water. AquaMats® allow those communities to be successful in a wide range of temperature and load conditions in the presence of oxygen and favorable water circulation patterns.

UltraWeave® Technology optimizes biological niche space to promote:

- Effective reduction of BOD and TSS,
- Phosphorus fixation/removal,
- Enhanced nitrification even in cold temperatures, and
- Reduction of excess minerals like copper and aluminum.

Design features include:

- High effective surface area: 200 m^2/m^3,
- Easy to install in new or existing pond systems without dewatering, substantial labor or heavy equipment, and
- Requires NO power or housing.

Figure 28. Courtesy of Meridian Aquatic Technology (2015) (Aquamats®).

Triplepoint Water Technologies, LLC (NitrOx™). Triplepoint Environmental's patent-pending NitrOx™ process was specifically designed to work within your existing infrastructure to cost-effectively remove ammonia from your lagoon effluent. It is a proven, time-tested solution to remove lagoon ammonia that can be installed at 2/3 the cost of existing nitrification options, while also as low maintenance as a lagoon.

Ammonia removal through nitrification will reliably occur with the proper conditions. The purpose of the NitrOx™ process is to control and optimize each of these conditions, fostering an ideal habitat for a specific set of bacteria called nitrifiers and super-concentrating them into a small, cost-effective, and highly efficient reactor. An increase in nitrifiers yields an increase in ammonia reduction. Nitrifiers grow on substrate or "media," meaning that in order to increase the quantity of these bacteria, you need to increase the surface area they grow on. The NitrOx™ reactor contains millions of small media, producing over 150 square feet of surface area per every cubic foot of volume. This provides for a dense concentration of nitrifiers, and allows the NitrOx™ process to be the most effective ammonia treatment solution in the smallest footprint, even at low temperatures.

As the water cools, nitrifiers consume ammonia more slowly. In the coldest winter months, wastewater can be too cold for the nitrifiers to effectively consume ammonia, so the NitrOx™ reactor automatically regulates its influent by gently heating the water to 39 to 42˚F with heat exchangers and geothermal heat, then maintaining this temperature with an insulated cover. Because this temperature increase is minor and only required a few months per year, the NitrOx™ reactor operates cost-effectively while promoting rapid ammonia reduction.

Low maintenance coarse bubble diffusers release large bubbles from a full-floor grid, transferring the oxygen nitrifiers need to complete their ammonia digestion. These coarse bubbles also distribute their high-energy mixing throughout the NitrOx™ reactor, agitating the millions of media so the bacteria, the oxygen, and the waste are continuously in contact, creating an ideal environment for rapid ammonia reduction. This same mixing also causes the media to collide with each other, the walls, and the piping sieves, constantly self-cleaning so you don't have to. Benefits of the NitrOx™ system include:

- Removes lagoon ammonia down to less than 0.1 mg/L ammonia;
- Maintains treatment in water temperatures near freezing;
- Cost-effectively retrofits your existing lagoons (2/3 the cost of alternatives);
- Low maintenance, simple operation; and
- Robust design that will last as long as your lagoon.

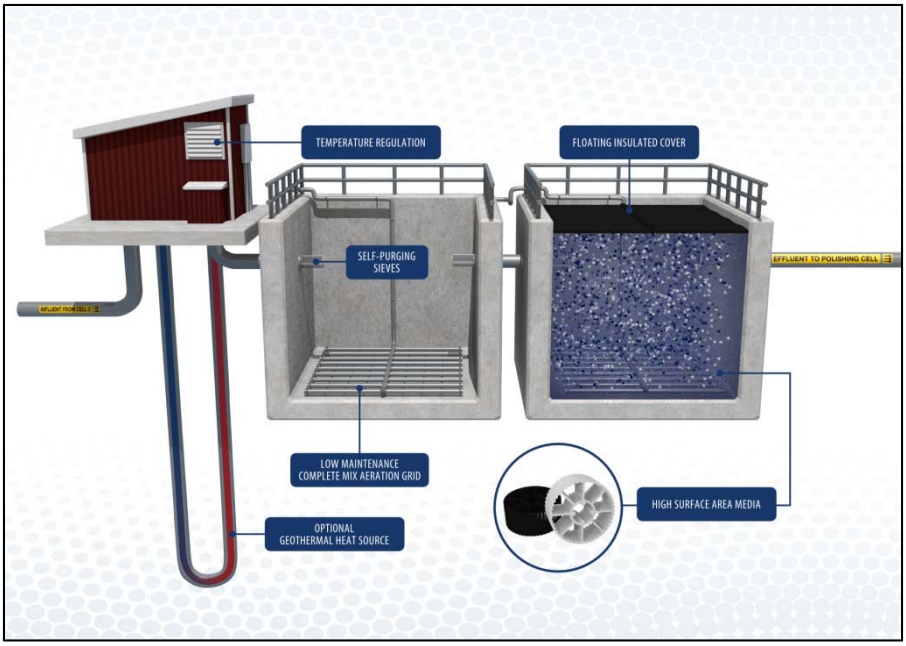

Figure 29. Courtesy of Triplepoint Water Technologies, LLC (2015) (NitrOx™).

ENTEX Technologies (BioWeb™). BioWeb™ is a knitted matrix fixed film media system used in Integrated Fixed-lm Activated Sludge (IFAS) systems and lagoon upgrades. Filaments form loops that stand out from the fabric, providing countless growth sites for attached biomass. The material, geometry and design mean excellent substrate distribution and oxygen diffusion.

BioWeb™ is a cost-effective solution for existing activated sludge plants that require additional treatment capacity or more advanced treatment. It is also an excellent choice for space efficient, high performance new plant design. It blends the process flexibility and advanced treatment benefits of activated sludge with the inherent stability of fixed film systems.

BioWeb™ modules are easily added directly into the aeration basins. The addition of attached growth biomass to the suspended biomass improves both performance and efficiency of a wastewater treatment plant. BioWeb™ allows more work to be done within existing space.

Figure 30. Courtesy of ENTEX Technologies (2015) (BioWeb™).

Recommendations

Fixed-film biological systems have been used widely in the treatment of wastewater, particularly in the attainment of secondary effluent standards and nitrification. Biofilm systems continue to draw significant research attention. While this review is not an all-encompassing documentation for the biofilm processes, it does provide an opportunity to reflect on what biofilm and hybrid biofilm systems may still have to offer the wastewater and environmental research and engineering community

Fixed film reactor systems can deliver a flexible, cost-effective, and easy-to-operate means to address current wastewater requirements and the expandability to meet future loads or more stringent discharge requirements within a compact design. However, upgrading existing facultative in Montana often requires a significant cost and skilled operators to achieve the full potential of the treatment options.

Some of the key features of fixed film process are provided below:

- It is stable under load variations, insensitive to temporary limitation and provides consistent treatment results.
- It generates low solids and requires no or minimum polymer for solid/liquid separation.
- It requires a small footprint. Typically it requires 1/3rd the space required for Activated Sludge Process.
- It requires a low capital cost and is comparable to cost of Activated Sludge Process. It is cheaper than the MBR process.
- Existing plants can be upgraded easily with many fixed film technologies. New fixed film plants can be upgraded to handle higher loads with no or minimum cost and construction.
- It is easy to operate, has automatic sludge wasting, has no sludge Return and no MLSS, and there no issue of media clogging.

- Capacity can be easily upgraded by simply increasing the fill fraction of biofilm carriers.
- No return activated sludge stream required.
- Actively sloughed biofilm automatically responds to load fluctuations.

References

Aeration Industries. Aire-O_2 product brochure.

An M. and K.V. Lo, 2001. Domestic wastewater treatment using immobilized sludge fluidized-bed reactors. J Environ Sci Health A Tox Hazard Subst Environ Eng. 2001 May; 36(5):819-31.

Brentwood Industries, Inc. 2015.

Brown, S., Get more for less: cost-effective solutions for lagoon conversions. Water Online: The Annual Wastewater Solutions Update, pp6-8.

Del Pozo R., and V. Diez. 2003. Organic matter removal in combined anaerobic–aerobic fixed-film bioreactors. Water Res 2003; 37:3561–8.

Environmental Dynamics International. 2015. IDEAL Solutions product brochure.

ENTEX Technologies. 2015. BioWeb product brochure.

FBC Technologies, Inc. 2015.

Galvez, J. M., M.A. Gomez, E. Hontoria, and J. Gonzalez-Lopez. 2003. Influence of hydraulic loading and air flowrate on urban wastewater nitrogen removal with a submerged fixed-film reactor. Journal of Hazardous Materials B101 (2003) 219–229.

Gullicks, H., H. Hasan, D. Das, C. Moretti, and Y-T. Hung. 2011. Biofilm fixed film systems. Water 2011, 3, 843-868; doi: 10.3390/w3030843.

Jenkins, A. M. and D. Sanders. 2012. Introduction to Fixed-Film Bio-Reactors for Decentralized Wastewater Treatment. Professional Development Advertising Section — Contech Engineered Solutions.

Kocadagistan, B., E. Kocadagistan, N. Topcu, and N. Demircioglu. 2005. Wastewater treatment with combined up-flow anaerobic fixed-bed and suspended aerobic reactor equipped with a membrane unit. Process Biochemistry 40 (2005) 177–182.

Lemna Technologies, Inc.2015. Lemtech Polishing Reactor product brochure.

Mendonca N.M., C.L. Niciura, E.P. Gianotti, and J.R. Campos. 2004. Full scale fluidized bed anaerobic reactor for domestic wastewater treatment: performance, sludge production and biofilm. Water Sci Technol. 2004; 49(11-12):319-25.

Meridian Aquatic Technology, LLC. 2006. AquaMats product brochure.

Nelson Environmental. 2015. SAGR product brochure.

Parker, D. S., M. Lutz, B. Andersson, and H. Aspegren. 1995. Effect of Operating Variables on Nitrification rates in Trickling Filters. Water Environ. Res., 67 (7), 1111.

Parker, D. S., T. Jacobs, E. Bower, D.W. Stowe, and G. Farmer. 1997. Maximizing Trickling Filter Nitrification Through Biofilm Control: Research Review and Full Scale Application. Water Sci. Technol., 36 (1), 255.

Tchobanoglous, G., F. Burton, and H.D. Stensel. 2003. Wastewater Engineering: Treatment and Reuse, 4th ed.; McGraw-Hill: New York.

Tisa, F., A. Raman, and W. Daud. 2014. Basic Design of a Fluidized Bed Reactor for Wastewater Treatment Using Fenton Oxidation. International Journal of Innovation, Management and Technology, Vol. 5, No. 2, April 2014.

Triplepoint Environmental. 2015. NitrOx Process product brochure.

U.S. Environmental Protection Agency (EPA Onsite Wastewater Treatment Systems Technology Fact Sheet 2 Fixed-Film Processes.

van Rijn, J., Y. Tal, and H.J. Schreier. 2005. Denitrification in recirculating systems: Theory and applications. Aquacultural Engineering 34 (2006) 364–376.

Wastewater Compliance Systems, Inc. 2015. Bio-Dome (Poo-Gloos) product brochure.

Zhu, J. and B. Rothermel. 2014. Everything you need to know about tricking filters. Clear Waters, Summer 2014, pp16-19.

4.2.3 Phosphorous Removal

General Description

Phosphorus, nitrogen and carbon are essential to the growth of microorganism in surface water; phosphorus discharge is often the limiting nutrient in surface water. Excess phosphorus (eutrophication) may lead to algal blooms or undesired growth in surface water. In order to limit phosphorus discharge from a lagoon, there are several removal mechanisms:

1. Chemical precipitation;
2. Enhanced biological uptake;
3. Tertiary treatment; and
4. Activated Alumina Adsorption.

Issues Addressed

Phosphorus occurs in wastewater lagoons primarily as phosphates (Pycha and Lopez 2003). Phosphates are organic, particulate (polyphosphate) and inorganic

(orthophosphates) and are a component of aquatic organisms for growth. Naturally occurring bacteria known as PAOs store excess phosphorous; some phosphorus is captured in the sludge and sediment in the bottom of the lagoon. However, standard facultative lagoons have minimal phosphorus attenuation and typically do not meet increased phosphorus removal regulations.

Stand-alone lagoon treatment (chemical precipitation and biological uptake) are proven methods to reduce phosphorus levels below 1 mg/L total phosphorus. Chemical precipitation typically includes metals salts such as aluminum sulfate (alum), ferric chloride or ferrous chloride. Other chemical additives include lime or polymers. Biological uptake may include harvesting of wetland plants, and aerated biofilm treatment with storage for high phosphorus waste streams. A general class of this treatment method is called EBPR. Additional end-of-lagoon treatment may be required to further reduce effluent levels; additional filtration after lagoon settling may reduce total phosphorus to 0.5 mg/L and chemical addition with a sand filter may reduce total phosphorus to levels below 0.02 mg/L (technologies such as BluePro™ and OPTAER®).

Design Variations

Chemical precipitation. In 2005, the Minnesota Environmental Science and Economic Review Board, (MESERB 2005) stated that "chemical precipitation is the only feasible process for phosphorus removal for lagoon systems". MESERB (2005) recommends the addition of aluminum sulfate (alum) in order to treat wastewater in the lower pH zones of the lagoon sediment. Chemical precipitation can be added as a batch system with motor boats, or with chemical feed equipment based on the flow of the wastewater influent.

According to Pycha and Lopez (2003), the factors that affect chemical precipitation include alkalinity, final pH of the wastewater, ionic constituents (sulfate, fluoride, and sodium), TSS, microorganisms, and mixing. Pycha and others (2003) states that "the optimal pH for phosphorus removal with alum ranges from 5.5 to 6.5, but in typical wastewaters, it ranges from 6.0 to 9.0… Ferric chloride is most effective in removing P when the pH ranges from 4.5 to 5.0, with typical values of 7.0 to 9.0."

Enhanced Biological Uptake. Enhanced biological uptake includes harvesting wetland plants and aerated biofilm infrastructure.

Farrell (2012) showed that Duckweed species *Lemna turionifera* and *W. borealis* may remove 113 mg-P/m^2 under 200 umol/m^2 sec light. This method requires sufficient land, residence time (study used 46 days), and continuous harvesting.

PAOs are a group of bacteria that, under certain conditions, facilitate the removal of large amounts of phosphorus from wastewater in a process, called EBPR. PAOs accomplish this removal of phosphate by accumulating it within their cells as polyphosphate. PAOs are by no means the only bacteria that can accumulate polyphosphate within their cells and in fact, the production of polyphosphate is a widespread ability among bacteria. However, the PAOs have many characteristics that other organisms that accumulate polyphosphate do not have, that make them amenable to use in wastewater treatment. Specifically, this is the ability to consume

simple carbon compounds (energy source) without the presence of an external electron acceptor (such as nitrate or oxygen) by generating energy from internally stored polyphosphate and glycogen. Most other bacteria cannot consume under these conditions and therefore PAOs gain a selective advantage within the mixed microbial community present in the activated sludge. Therefore, wastewater treatment plants that operate for enhanced biological phosphorus removal have an anaerobic tank (where there is no nitrate or oxygen present as external electron acceptor) prior to the other tanks to give PAOs preferential access to the simple carbon compounds in the wastewater that is influent to the plant.

Post-Lagoon treatment and filtration (tertiary treatment). Limited peer-reviewed articles were available for post-lagoon treatment and filtration. Several technologies are available and include Blue PRO® (www.bluewater-technologies.com) and OPTAER™ (www.nelsonenvironmental.com). Both continuous backwash gravity sand filters; Blue PRO® includes iron addition for adsorption onto hydrous ferric oxide. This methodology requires a tertiary treatment step and operator involvement. A photograph from OPTAER™ continuous backwash sand filters following a lagoon is included below.

Figure 31. Blue PRO® is a fluidized sand bed with iron chemical addition. Removal rates on a sewage lagoon in Kellogg, Idaho lowered TP from 2.5 mg/L to 0.020 mg/L.

Activated Alumina Adsorption. Several communities have activated alumina in packed beds for the treatment of potable water (fluoride, arsenic, selenium, uranium, beryllium, and natural organic matter) (Bureau of Reclamation 2012). Lab scale testing has been completed to determine the ability of activated alumina to polish wastewater effluent from tertiary treatment (Kang and Lee 2012). Lab scale column tests were used with synthesized phosphate solution (2 mg/L) with metal-impregnated activated alumina balls. Calcium impregnated activated alumina removed approximately 72% of the phosphate compared to the next best impregnated metal (aluminum, 55% removal).

The influent solution from the tertiary treatment had 0.22 mg/L total phosphorus and 0.14 mg/L phosphate and was fed to a continuous lab scale reactor with calcium impregnated activated alumina. Tertiary treatment included a membrane bioreactor and a coagulated/dissolved air floatation tertiary treatment. A continuously fed lab scale column test was measured in bed volumes until breakthrough was detected, or approximately 4,000 bed volumes. This lab scale study shows the feasibility of treating high concentration phosphates (2 mg/L) with synthetic water and the feasibility of

treating low concentration phosphorous from tertiary wastewater treatment (0.22 mg/L). Large scale use in wastewater treatment was not found.

Operations and Maintenance Issues

Chemical Precipitation. Pycha and Lopez (2003) reviewed batch facultative lagoon usage of chemical precipitation from Canada (20 lagoon systems) and Minnesota (11 lagoon systems) into the secondary cell of the lagoon. Operations include delivery of alum in either liquid form (tanker trunk) or in bags (dry form) and storage on-site. The operator mixes the alum with water to form a solution; the solution is applied to the lagoon with a motor boat, storage tank, and chemical feed equipment. The alum is either fed into a prop-wash and mixed with the outboard motor into the lagoon or is sprayed onto the wastewater surface. The alum is dosed into the lagoon based on the results of a wastewater sample and dosage chart or based on the operator's past experience. The boat travels across the lagoon in a grid-like manner. The floc is given a minimum amount of time to settle, and then the wastewater is seasonally discharged from the secondary cell.

Pycha and Lopez (2003) reviewed 26 municipal lagoon systems in Michigan that include seasonal discharge (2 times per year) and continuous discharge systems. All of the facilities relied on the mixing chamber between the lagoon cells and clarifier to dose the chemical additive.

Maintenance issues for all chemical precipitation lagoons in cold weather regions (Canada, Michigan, and Minnesota) include: 1) Seasonal algae blooms (increases TSS in effluent); 2) Mixing of surface wastewater, algae and duckweed (re-suspension of precipitated solids, increase BOD and increase TSS); 3) Handling, storage and mixing of chemicals; and 4) Reliance on a dose chart or operator experience rather than representative sampling results. There were minimal maintenance problems associated with sludge as the accumulated amounts were consistent with buildup in the primary lagoons.

Enhanced Biological Uptake. Promoting growth of duckweed and continuous harvesting requires labor during the growing season, a temperate climate and disposal/re-use of the harvested plant material. Ferrell (2012) includes options for the harvested biomass included anaerobic digestion, fermentation for ethanol production, or as an animal feed. Applicability of biomass uptake of phosphorus is unknown in Montana.

Fixed bio-film removal of phosphorus requires alternating anaerobic and aerobic environments. PAOs uptake orthophosphate during the aerobic phase and release orthophosphate during the anaerobic phase. Johnson (2011) states that management of the sludge to prevent re-release of phosphorus into the effluent is the biggest problem. Denitrifying PAOs produce less sludge; thus simultaneous nitrogen and phosphorous removal is considered more economical. Johnson (2011) markets the Poo-gloo™ which requires the addition of fixed film aeration devices, air cycling in the infrastructure and diversion/storage/re-use of phosphorous laden effluent.

Tertiary Treatment. A fluidized bed filter (or continuous backwash gravity sand filter) would be a best fit for a continuously discharging lagoon as there is significant capital infrastructure. Tertiary equipment is available in free-standing units, in-ground concrete cells, or skid-mounted systems. Adsorbed phosphorous within the media, or the concentrated phosphorous waste stream generated during back-wash would require disposal and/or storage.

Activated Alumina Adsorption. Activated alumina adsorption requires minimal operator involvement. Wastewater would flow through packed beds of activated alumina until analytic tests show a breakthrough of phosphorus in the lead column. Typically adsorption is set up with a lead and lag column (or contact bed). The limitation of adsorption resins is that the resin must be replaced upon breakthrough or regenerated. Regeneration may require caustic or acid wash and may generate a hazardous waste. Storage of the caustic or acid must follow safe precautions with spill prevention (Bureau of Reclamation 2010).

Relative Costs

Chemical Precipitation. The cost of chemical precipitation equipment was obtained from Williams (2012) in which "chemical addition and settling units were added to the treatment schematic following the lagoon(s) and prior to disinfection and discharge". Cost estimates are in January 2012 dollars with 20% engineering fees and 10% contingency. For 24 facilities that would be permitted to have an effluent phosphorus concentration of 0.5 to less than 1.0 mg/L would range from $164,000 to $580,000. The average cost was $484,000.

EBPR – Fixed Bio-film. According to Johnson (2011) there are no commercially deployed in-situ wastewater lagoon products. This is a new development in lagoon treatment; as recent as 2007 in a report to MDEQ, "chemical addition is the recommended phosphorous reduction process at plants without suspended-growth activated sludge systems to which EBPR may be retrofitted". Poo-gloos™ retrofit systems are $100,000 to $500,000 with electrical costs to run the aerator, and alarming mechanisms. Poo-gloos™ have air systems that have a 10 to 15 year design life and according to Johnson (2011) "conventional chemical precipitation methods to remove phosphorous from wastewater streams have an associated O&M cost factor much higher for a comparable flow rate, excluding substantial costs for disposing the large volumes of the unusable sludge that is created in the process". Poo-gloos™ divert high-phosphorus waste to a dedicated pond for use as fertilizer. If the aeration to the bio-film dome is continuous, then ammonia, nitrogen compounds would be removed. Phosphorus removal requires the intermittent on/off cycling and diversion of the sloughed high-phosphorus bio-mass.

Tertiary Treatment. The capital and operator costs for tertiary treatment are not provided; however they exceed chemical precipitation and EBPR costs. Blue PRO® states that it is "…the leading technology for phosphorus reduction to any level. Whether the targeted phosphorus discharge limit is 10 mg/L P or as low as 0.01 mg/L, Blue PRO® can provide reductions in chemical usage, equipment footprint, and associated operations and maintenance costs over alternative technologies. Blue PRO® is the most effective and most inexpensive tertiary treatment solution where additional considerations are

needed, such as denitrification or metals removal." The comparison is most likely referring to other tertiary treatment alternate technologies.

Activated Alumina. Full scale activated alumina for wastewater treatment was not found. Relative costs were not available as a wastewater polishing step.

Constructability

Constructability is different based on the required concentration of phosphorus in the effluent. The technologies ramp up in difficulty – from delivering chemical via boat to an existing lagoon, to installation of filtration process units or adsorption process units. Additional storage ponds and available land for storage/reuse may cause the selection of EBPR systems to be less favorable.

Recommendations

The selection of a phosphorus removal technology is strongly tied to the regulatory environment and the existing BOD treatment. Effluent levels from an activated sludge or fixed film lagoon may meet regulatory compliance limits. Technologies such as the Poo-gloos™ are potential upgrades that could be made to Montana facultative lagoons to achieve high total phosphorus removal levels.

References

Ferrell, Jonathan B. 2012. Duckweed Uptake of Phosphorous and Five Pharmaceuticals: Microcosms and Wastewater Lagoon Studies, Utah State University, January 2012 Available at: http://digitalcommons.usu.edu/cgi/viewcontent.cgi?article=2206&context=etd&sei-redir=1&referer=http%3A%2F%2Fwww.bing.com%2Fsearch%3Fq%3Dbiological%2520uptake%2520efficiency%2520lagoon%26pc%3Dcosp%26ptag%3DAA2F24ADA02%26form%3DCONOMX%26conlogo%3DCT3210127#search=%22biological%20uptake%20efficiency%20lagoon%22

Hartman, P. and J. Cleland. 2007. Wastewater Treatment Performance and Cost Data to Support an Affordability Analysis for Water Quality Standards, Prepared For: Michael Suplee, Ph.D., MDEQ, Helena MT Prepared by ICF International, Lexington, Massachusetts, May 31, 2007. Available at: file:///C:/Users/kdraper/Downloads/Nutrient%20Treatment%20Cost%20Data_053107%20(1).pdf

Johnson, Kraig. 2011. Rural wastewater treatment lagoon enhancement with dome shaped submerged bio-film devices. Wastewater compliance systems, Inc.

Kang, S.C and B.H. Lee. 2012. Removal of Phosphorous in Wastewater by Ca-Impregnated Activated Alumina. Environmental Engineering Research, pages 197-203, December 14, 2012. Available at: www.eeer.org/upload/eer-17-4-197-.pdf

MESERB. 2005. Wastewater Phosphorus Control and Reduction Initiative. Prepared for the Minnesota Environmental Science and Economic Review Board. Prepared by HydroQual, Inc., Mahwah, NJ, in association with H. David Stensel, Ph.D., P.E., University of Washington, Seattle, WA. Project Number MESE0001. April 2005. Available at: http://www.meserb.org/wp-content/uploads/2009/12/mesereport22apr05.pdf.

Pycha, C. and E. Lopez. 2003. Municipal Wastewater Lagoon Phosphorus Removal, Prepared by Environmental Engineers, Technical Support Section, Water Compliance Branch, U.S. EPA 5WCT-15-J, Chicago, IL. Available at: http://www.lagoonsonline.com/phosphorous.htm

USDA. 2010. Rural Wastewater Treatment Lagoon Enhancement with Dome Shaped Submerged Bio-Film Devices, Project Number UTAK-2010-00232. United States Department of Agriculture Research, Education & Economics Information System. Available at: http://www.reeis.usda.gov/web/crisprojectpages/0221807-rural-wastewater-treatment-lagoon-enhancement-with-dome-shaped-submerged-bio-film-devices.html

U.S. Department of the Interior, Bureau of Reclamation. 2010.

U.S. Department of the Interior, Bureau of Reclamation. 2012. Activated Alumina Fact Sheet, September 20, 2012. Available at: http://www.usbr.gov/pmts/water/publications/reportpdfs/Primer%20Files/07%20-%20Activated%20Alumina.pdf

Williams, M.B. 2012. Cost of Phosphorus Removal at Wisconsin Publically-Owned Treatment Works. Prepared by Mark Williams, PE, Professional Consultant, Wisconsin Department of Natural Resources, Division of Customer and Employee Services, Bureau of Community Financial Assistance, Environmental Loans Section, December 2012. Available at: http://dnr.wi.gov/topic/surfacewater/documents/MarkWilliamsReport.pdf

4.2.4 Alternate Carbon Sources

General Description

Two of the most commonly measured constituents of wastewater BOD and total nitrogen. Wastewater is composed of a variety of inorganic and organic substances. Organic substances refer to molecules that are based on carbon and include fecal matter as well as detergents, soaps, fats, greases, and food particles (especially where garbage grinders are used). These large organic molecules are easily decomposed by bacteria in the wastewater treatment system. However, oxygen is required for this process of breaking large molecules into smaller molecules and eventually into carbon dioxide and water. The amount of oxygen required for this process is known as the BOD. The five-day BOD, or BOD_5, is measured by the quantity of oxygen consumed by microorganisms during a five-day period, and is the most common measure of the amount of biodegradable organic material in, or strength of, sewage. Sewage high in BOD can deplete oxygen in receiving waters, causing fish kills and ecosystem changes.

However, BOD content of sewage is also important for wastewater treatment systems, including facultative lagoons. Some BOD is removed in the anaerobic zone of the lagoon by anaerobic digestion and by solids which settle to the bottom of the lagoon. However, much of the BOD present in sewage (especially detergents and oils) remains in the water until enough oxygen is present to facilitate the bacterial processes to remove the BOD. This same process also converts the ammonia and organic nitrogen to nitrate,

resulting in wastewater with high ammonia concentration and virtually no BOD. Too much BOD can cause excessive growth of bacteria in the lagoon.

BOD is fairly easy to remove from sewage by providing a supply of oxygen during the treatment process; the oxygen supports bacterial growth which breaks down the organic BOD. Most enhanced treatment units incorporate some type of unit which actively oxygenates the sewage to reduce BOD. Reduction of BOD is a relatively easy and efficient process, and results in sewage of low BOD. It is also important to note that BOD serves as the food source for the denitrifying bacteria which are needed in systems where bacterially-mediated nitrogen removal takes place. In these situations BOD is desired, as the nitrification/denitrification process cannot operate efficiently without sufficient BOD to support the growth of the bacteria which accomplish the process.

As discussed previously in this document, the reduction of total nitrogen is a two-step process. The first step, the conversion of ammonia to nitrite and then to nitrate, is called nitrification. It is important to note that nitrification requires and consumes oxygen, and is carried out simultaneously with the BOD removal process. The nitrification process is mediated by the bacteria, which require an aerobic environment for growth and metabolism of nitrogen.

The second step of the process, the conversion of nitrate to nitrogen gas, is referred to as de-nitrification. This process is also mediated by bacteria. For de-nitrification to occur, the DO level must be at or near zero. The bacteria also require a carbon food source for energy and conversion of nitrogen. The bacteria metabolize the carbonaceous material or BOD in the wastewater as this food source, metabolizing it to carbon dioxide. This in turn reduces the BOD of the sewage, which is desirable. However, if the sewage is already low in BOD, as is the case in the nitrification/de-nitrification process, the carbon food source will be insufficient for bacterial growth and de-nitrification will not proceed efficiently. To overcome this problem, the system must incorporate an external source of organic carbon to achieve nitrate removal.

Issues Addressed

The main issue addressed by adding BOD to a wastewater treatment system is total nitrogen. In typical wastewater, if conditions are favorable to reduce BOD, they are also sufficient to carry out the nitrification process. Once all the BOD and ammonia are oxidized, there is insufficient food for the denitrifying bacterial to effectively treat the nitrate rich water. To achieve greater total nitrogen removal, external carbon sources must be added.

Design Variations

The effective reduction of total nitrogen from a wastewater system requires denitrification. The three things required for denitrification are nitrified water, anaerobic conditions, and organic carbon. Since the BOD was removed during nitrification, organic carbon is often the limiting constituent for denitrification. In general terms, there are two main for forms of organic carbon that are used for denitrification: solid and liquid.

Liquid Organic Carbon. Numerous liquid carbon sources are used to meet the organic carbon needs for denitrification. The most widely used liquid carbon sources are methanol, ethanol, acetate, acetic acid, glycerol, molasses, and proprietary formulations like MicroC™. The following paragraphs provide details of the specific liquid sources.

- ***Methanol.*** Methanol is the most common external carbon source used in denitrification processes because it is widely available and cost-effective. Methanol does have several drawbacks. Methanol is highly flammable (flashpoint of 12°C [54°F]). Many phosphorus-accumulating organisms cannot use methanol, therefore, if a treatment plant needs carbon for denitrification and volatile fatty acids (VFAs) for biological phosphorus removal, methanol cannot be used. Methanol may not be the most efficient source of carbon for denitrification in systems with brief hydraulic residence times (HRTs) and solids retention times. Finally, the cost and availability of methanol has the potential for wide fluctuations.
- ***Ethanol.*** The use of ethanol as the carbon source for denitrification is becoming a widely used alternative to the more commonly used methanol both with respect to economy and process flexibility. Pure methanol is now often more expensive than pure ethanol, and ethanol can be obtained as an inexpensive waste product from chemical and pharmaceutical industries. Also, methanol has been shown to select for a highly specialized denitrifying population consisting of facultative methylotrophs (Claus and Kutzner 1985; Timmermans and van Haute 1983).
- ***Acetate.*** Acetate is known as an easily degradable carbon compound for many bacteria. Isaacs and others (1994) showed that nitrogen removal was enhanced immediately after acetate addition (Isaacs and Henze 1995).
- ***Acetic acid.*** Acetic acid (vinegar) is often used as the VFA of choice for biological phosphorus removal, but it is also an efficient carbon source for denitrification processes. Acetic acid is less volatile than methanol, but concentrated acetic acid solutions have a relatively low flashpoint and a freezing point of 17°C (62°F); therefore, if concentrated solutions are used, the treatment facility must meet code requirements for storage of flammable liquids and include measures to avoid freezing.
- ***Glycerol.*** Glycerol is a byproduct of biodiesel production, created during the transesterification process. Biodiesel is a cleaner-burning, biodegradable fuel made from renewable resources, such as vegetable oils and animal fats. When reacted with alcohol (usually methanol), such fats and oils generate biodiesel and a crude glycerol, which contains about 20% water. As biodiesel gains popularity, more glycerol will be produced.
- ***Molasses.*** Molasses, often referred to as sugar industry waste, has been evaluated and used in the wastewater treatment industry. Tatsuki Ueda and others (2006) studied biological denitrification using sugar industry wastes (final molasses) as a carbon source with charcoal pellets as supporting media for denitrifying bacteria. The study demonstrated that final molasses could be used as a carbon source for biological denitrification.
- ***MicroC™.*** MicroC™ and MicroC G™ are made by Environmental Operating Solutions (Bourne, Massachusetts). MicroC™ is a proprietary mix of

agriculturally derived compounds and 5% methanol, which lowers the freezing point and reduces the viscosity (deBarbadillo 2008). MicroC™ is a light green liquid compound with a mild alcohol odor containing agricultural products and methanol (5% w/w). Its bulk density, specific gravity, and viscosity are 1.18 g/cm^3, 1.18, and 0.0164 kg/m-s, respectively. The compound is soluble in water, and its pH at 25 ^{o}C is 5.8. MicroC™ is stable under normal conditions and has a freezing point of 220 ^{o}C, which avoids storage issues during cold seasons (Cherchi et al. 2009).

Solid Organic Carbon. Until recently, liquid carbon sources such as methanol and molasses have been used to stimulate microbial denitrification (Rabah and Dahab 2004; Modin et al. 2007; Ginige et al. 2009). However, liquid carbon sources are costly, require sophisticated process control, and pose the risk of overdose. Accordingly, a great deal of attention has recently been focused on solid carbon sources (Saliling et al. 2007; Robinson-Lora and Brennan 2009). The most widely used solid carbon sources include: woodchips, sawdust, corn husks, wheat straw, and biodegradable plastic.

The longevity of nitrate removal in denitrification systems is important to their actual application. In most cases, only slow rates of solid carbon sources decomposition are needed to support nitrate removal because there is generally a large amount of carbon relative to the nitrate inputs. Consequently, the appropriate solid carbon sources should have a low release of soluble nitrogen components, slow degradation rates and high denitrification rate (Zhang et al. 2012).

Jing and others (2010), examined using walnut shells as a solid carbon source. The experiment results indicated that walnut shell could achieve a great nitrate removal efficiency of 80%, and the concentration of nitrates in effluent was below 3 mg/L. During the experiment, pH and water temperature had been almost constant, and the concentration of nitrite in effluent was low. One of the main factors affecting this reaction was HRT, and the best HRT was 24 hours. The denitrification of groundwater using walnut shell as solid carbon source conformed to the zero-order kinetics equation. The main components of walnut shell are lignin, cellulose, and hemicellulose which can be consumed by denitrifying bacteria.

Boley and others (2000) evaluated the use of biodegradable polymers for the denitrification of aquaculture wastewater. They found that in contrast to conventional treatment units, denitrification with biodegradable polymers presented here is a simple process. Microorganisms use the biopolymer in form of pellets as biofilm carrier and simultaneously as water insoluble carbon source for denitrification. The system with biodegradable polymers does not require an external dosing of soluble organic substrate as the polymer itself acts as biofilm carrier and organic carbon source.

Capodici and others (2014) evaluated four solid carbon sources and their use in permeable reactive barriers to remove nitrate from groundwater. According to Capodici and others (2014), in the absence of oxygen, solid organic carbon is oxidized, by heterotrophic bacteria naturally present in soil, in compliance to anoxic metabolism by using nitrates bound oxygen. This reaction converts nitrates in nitrogen gas. Organic substrates capable to sustain this reaction are various and easily recoverable (e.g.

sawdust, cotton, woodchips etc.); thus, several carbon source has been already tested. The present paper reports the results of batch test carried out on four organic substrates used to promote biological denitrification; in details the organic matters tested were: sawdust, pine bark, cork, and olive pomace. All organic matters tested provided good results, in terms of removal efficiency.

Shao and others (2008), used rice husks as a carbon source and biofilm carrier for denitrification. During the experiment, they investigated rice husks as the sole carbon source as well as biofilm carrier in the biological denitrification of wastewater in up-flow laboratory reactors. Fast startup of the reactor and high nitrate removal efficiency were observed. Flow rate and nitrate concentration of the influent were observed to have a significant effect on nitrate removal efficiency. The reactor had the ability to accommodate a wide range of pH (6.5 to 8.5) and DO (1.5 to 4mg/L). A time-dependent decrease in nitrate removal efficiency was observed after 72 days of operation. The addition of new rice husk brought about a rapid increase of nitrate removal efficiency. Results showed that rice husks can be an economical and effective carbon source for the nitrate removal process (Shao et al. 2008).

Operations and Maintenance Issues

Adding organic carbon for denitrification in a facultative lagoon could cause several operations and maintenance issues. The denitrification process produces additional bio-solids that could require more frequent cleaning and could cause short-circuit of the pond flow scheme. Also, depending on the type of organic carbon material, the distribution and storage of the material could cause additional operations and maintenance activities.

Relative Costs

The costs associated with incorporating an external carbon source for denitrification are tied to other operational and mechanical upgrades for facultative lagoons, thus costs could range from moderate to high, depending on the associated upgrades. For example, in order for denitrification to occur, the water has to be nitrified first, thus the likely need for aeration upgrades would be necessary prior to adding external carbon.

Constructability

The actual constructability of adding an external source ranges from very simple to very complex depending on the type of carbon source selected. For example, a liquid carbon source, such as methanol, would require a dosing and mixing system, a secure storage area, and a trained operator. A solid carbon source would require very little in terms of distribution, but would require occasional material replacement, depending on the mass of material added per dose.

New/Innovative Approaches

The use of solid carbon sources appears to be the current hot spot for denitrification research. The liquid components have been around for quite some time and require complicated dosing and mixing apparatuses. Anderson (2012), patent number US 8,241,717B1, describes the present invention concerns a carbon-based biofilm carrier in a hollow shape constructed of a combination of carbon-based materials such as: saturated fatty acids, saturated fatty alcohols, sawdust, cellulose based acetate

polymers, or biodegradable plastics, and formed traditional forming methods. Carriers are formed in a hollow shape to provide maximum surface area and protection during aerobic and anaerobic biofilm processes.

Recommendations

It is recommended that external carbon sources, specifically solid carbon sources, be evaluated on a site-by-site basis to determine if they could be combined with additional technologies to enhance total nitrogen removal. Complimentary technologies include: aeration, floating covers, floating islands, and other ecological technologies.

References

Anderson, S. M. 2012. Carbon-based biofilm carrier. United States Patent US 8,241,717B1.

Boley, A., W.-R. Muller, and G. Haider. 2000. Biodegradable polymers as solid substrate and biofilm carrier for denitrification in recirculated aquaculture systems. Aquacultural Engineering 22 (2000) pp75-85.

Capodici, M., C. Morici, and G. Viviani. 2014. Batch Test Evaluation of Four Organic Substrates Suitable for Biological Groundwater Denitrification, Chemical Engineering Transactions, VOL. 38, 2014, pp43-48.

Cherchi, C., A. Onnis-Hayden, I. El-Shawabkeh, and A.Z. Gu. 2009. Implication of using different carbon sources for denitrification in wastewater treatments. Water Environment Research, Volume 81, Number 8, pp788-799.

Claus G. and H.J. Kutzner. 1985. Denitrification of nitrate and nitric acid with methanol as carbon source [J]. Appl Microbiol Biotechnol, 22: 378–381.

deBarbadillo, C., J. Barnard, S. Tarallo, and M. Steichen. 2008. Got carbon? Widespread biological nutrient removal is increasing the demand for supplemental sources. WE &T magazine, January 2008, pp49-53.

Ginige, M., J. Bowyer, L. Foley, J. Keller, and Z. Yuan. 2009. A comparative study of methanol as a supplementary carbon source for enhancing denitrification in primary and secondary anoxic zones. Biodegradation 20 (2), 221–234.

Isaacs S H, M. Henze, H. Soeberg et al. 1994. External carbon source addition as a means to control an activated sludge nutrient removal process. [J]. Wat Res, 28(3): 511–520.

Isaacs S. H. an d M. Henze. 1995. Controlled carbon source addition to an alternating nitrification-denitrification wastewater treatment process including biological P-removal [J]. Wat Res, 29(1): 77–89.

Jing, L., H. Wang, T. Jiang, D. Li, H. Feng, and M. Chen, 2010. Denitrification of groundwater using walnut shell as solid carbon source. Conference on Environmental Pollution and Public Health.

Modin, O., K. Fukushi, and K. Yamamoto. 2007. Denitrification with methane as external carbon source. Water Research 41 (12), 2726–2738.

Rabah, F. K. J. and M.F. Dahab. 2004. Nitrate removal characteristics of high performance fluidized-bed biofilm reactors. Water Research 38 (17), 3719–3728.

Robinson-Lora, M. A., and R. A. Brennan. 2009. "Efficient Metal Removal and Neutralization of Acid Mine Drainage by Crab-Shell Chitin under Batch and Continuous-Flow Conditions," Bioresource Technology 100(21): 5063–71.

Saliling, W. J. B., P.W. Westerman, and T.M. Losordo. 2007. Wood chips and wheat straw as alternative biofilter media for denitrification reactors treating aquaculture and other wastewaters with high nitrate concentrations. Aquacultural Engineering 37 (3), 223–233.

Shao, L., Z.X. Xu, W. Jin, and H.L. Yin. 2008. Rice husk as carbon source and biofilm carrier for water denitrification. Polish J. of Environ. Stud. Vol. 18, No. 4 (2009), pp693-699.

Tatsuki Ueda, Yoshiyuki Shinogi, and Masaru Yamaoka. 2006. Biological nitrate removal using sugar-industry wastes. Paddy Water Environ, 4; 139-144.

Timmermans P. and A. Van Haute. 1983. Denitrification with methanol, fundamental study of the growth and denitrification capacity of Hyphomicrobium sp. [J]. Wat Res, 17(10): 1249–1255.

Zhang, J., C. Feng, S. Hong, H. Hao, and Y. Yang. 2012. Behavior of solid carbon sources for biological denitrification in groundwater remediation. Water Science & Technology, 65.9, pp1696-1704.

Srinu Naik, S. and Y. Pydi Setty. 2012. Biological denitrification of wastewater-a mini review on carbon source. International Conference on Chemical, Environmental Science and Engineering (ICEEBS'2012) July 28-29, 2012 Pattaya (Thailand).

Playchoom, C., W. Pungrasmi, and S. Powtongsook. 2010. Effect of carbon sources and carbon/nitrogen ratio on nitrate removal in aquaculture denitrification tank. 2010 International Conference on Biology, Environment and Chemistry IPCBEE vol.1, pp307-311.

Martinsa, C.I.M., E.H. Edinga, M.C.J. Verdegema, L.T.N. Heinsbroeka, O. Schneiderc, J.P. Blanchetond, E. Roque d'Orbcasteld, and J.A.J. Verretha. 2010. New developments in recirculating aquaculture systems in Europe: A perspective on environmental sustainability. Aquacultural Engineering, November 2010, Volume 43, Issue 3, pp83-93.

Hamlin, H.J., J.T. Michaels, C.M. Beaulaton, W.F Graham, W. Dutt, P. Steinbach, T.M. Losordo, K.K. Schrader, and K.L. Main. 2007. Comparing denitrification rates and

carbon sources in commercial scale upflow denitrification biological filters in aquaculture. Aquacultural Engineering 38 (2008) pp79-92.

Godini, H., A. Rezaee, A. Jafari, and S. H. Mirhousaini. 2010. Denitrification of wastewater containing high nitrate using a bioreactor system packed by microbial cellulose. International Science Index International Journal of Vol: 4 No: 2, pp239-243.

U.S. EPA. 2013. Wastewater Treatment Fact Sheet: External Carbon Sources for Nitrogen Removal.

Ying Xu, Tian-Lei Qiu, Mei-Lin Han, Jun Li, Xu-Ming Wang., 2011, Heterotrophic Denitrification of Nitrate- Contaminated Water Using Different Solid Carbon Sources, Procedia Environmental Sciences, 10, 72- 77.

4.2.5 Pathogens/Disinfection

General Description

The United States EPA in 1973 adopted regulations for secondary treatment in response to the Clean Water Act. The regulations included fecal coliform (FC) bacteria; however in 1976 regulations for FC were replaced with state site-specific water quality standards. Montana does not have a FC limit in MDEQ Circular DEQ-7; but regulations are in place for *Escherichia coli* (*E. coli*). Montana Pollutant Permit Discharge Elimination System (MPDES) will typically identify an *E. coli* limit that may be higher in winter months (November to March) and lower in summer months (April to October) to allow for drinking and swimming possible uses.

Typical lagoon disinfection methods include: chlorination, chlorination/dechlorination, UV disinfection, and ozone. An overview of the issues of chlorine and ozone disinfection is included, as well as a detailed analysis of the applicability of UV radiation or coagulation/filtration for facultative lagoons.

Issues Addressed

Facultative lagoons provide for removal of pathogens due to the facultative processes; however additional removal may be necessary based on regulatory limits.

Metcalf and Eddy (2003) show removal or destruction of bacteria by the following treatment processes: plain sedimentation (25 to 75%), chemical precipitation (40 to 80%), trickling filters (90 to 95%), activated sludge (90 to 98%), and chlorination of treatment wastewater (98 to 99.999%). According to H&S Environmental (2003), the residence time of facultative lagoons and exposure to solar radiation increase fecal coliform and other pathogen removal over mechanical plants. Common disinfection methods include chemical agents (such as chlorine or ozone) and radiation (UV light). Metcalf and Eddy (2003) break down the factors that influence the selection of disinfection products to six tiers: 1) contact time, 2) disinfectant concentration, 3) intensity of UV light, 4) temperature, 5) types of organisms, and 6) nature of the suspending liquid. With respect to facultative lagoons for small Montana communities,

it appears that issues include: permit compliance, cost, safe storage and handling of chemicals, and operator involvement.

The principal compounds used for chlorine disinfection at wastewater treatment plants include: chlorine, sodium hypochlorite, calcium hypochlorite, and chlorine dioxide. The use of chlorine has several concerns: 1) chlorine is toxic and may pose health risks to treatment plant operators, 2) there is risk during transportation, 3) spill prevention and containment are required for storage, 4) chlorine reacts with organic constituents in wastewater to produce odorous compounds and/or carcinogenic or mutagenic byproducts, 5) residual chlorine in effluent is toxic to aquatic life, and 6) unknown long term effects of discharge of chloro-organic compounds to the environment.

Byproducts are formed as a result of free chlorine and humic acids. Metcalf and Eddy (2003) state that the principal method of limiting the formation of disinfection byproducts (DBP) is avoiding the direct addition of free chlorine dechlorination methods such as reacting the residual chlorine with a reducing agent or adsorbing on activated carbon. Oxidation with ozone instead of chlorine also causes disinfection byproducts. Control of byproduct formation from ozone includes filtration with a biologically active filter or carbon column; if bromine is present and brominated disinfection byproducts are present, then an alternative means of disinfection is recommended (UV radiation).

Per Margensin and Schinner (1999), additional chlorine is required to treat lagoon wastewater that is colder than 15 degrees Celsius. The additional chlorine increases the disinfection byproduct risk. Their recommendation is to use coagulation/filtration for facultative lagoons.

Design Variations

Chlorine addition is a prominent disinfection design for lagoon effluent. The design of UV disinfection and coagulation/filtration are discussed in this report.

UV Disinfection. Ultraviolet disinfection is effective for pathogen control of lagoon effluent (H&S Environmental 2003). UV disinfection infrastructure includes the UV lamps, quartz sleeves to house the UV lamp, the supporting structure to hold the quartz sleeves and lamp, the ballasts used to supply regulated power to the lamp, and the power supply. The wastewater most often contacts the submerged lamps; however, non-contact reactors are available. Fully engineered systems with UV radiation infrastructure and open channel structures (concrete or plastic) are available for purchase to minimize engineering design. One example of these systems is Xylem's new WEDECO TAK 55 Smart UV light disinfection system (www.filstep.com).

Coagulation/filtration. Finch and Smith (1986), showed 99.9% reduction on fecal coliforms from lagoon wastewater by batch treatment with aluminum sulfate (alum). Pycha and Lopez (2003) reviewed batch facultative lagoon usage of chemical precipitation from Canada (20 lagoon systems) and Minnesota (11 lagoon systems) into the secondary cell of the lagoon. Operations include delivery of alum in either liquid form (tanker trunk) or in bags (dry form) and storage on-site. The operator mixes the alum with water to form a solution; the solution is applied to the lagoon with a motor

boat, storage tank, and chemical feed equipment. The alum is either fed into a prop-wash and mixed with the outboard motor into the lagoon or is sprayed onto the wastewater surface. The alum is dosed into the lagoon based off of a wastewater sample and dosage chart or based off of lagoon operator experience. The boat travels across the lagoon in a grid-like manner. The floc is given a minimum amount of time to settle, and then the wastewater is seasonally discharged from the secondary cell.

Pycha and Lopez (2003), reviewed 26 municipal lagoon systems in Michigan that include seasonal discharge (2 times per year) and continuous discharge systems. All of the facilities relied on the mixing chamber between the lagoon cells and clarifier to dose the chemical additive. Note that all lagoons in Michigan with alum treatment have additional disinfection systems and a clarifier to settle the flocculent.

Operations and Maintenance Issues

Ultra Violet Radiation. UV Radiation must penetrate to pathogens; thus effectiveness decreases on sloughing of biofilms, clumping of material, or high TSS/low UV transmissivity. Filtration is often required upstream of UV radiation; however, an engineering firm KWL Consulting Engineers presented a study on the Fort Nelson's, British Columbia, Canada, municipal lagoon in which pathogen removal was met with UV radiation without filtration. This project required comprehensive monitoring of wastewater effluent during conceptual and pre-design stages of the study. Additional problems associated with UV radiation include: 1) no immediate measurement of success; 2) no residual effect; 3) less effective than chlorine for some viruses, spores, etc.; 4) energy-intensive; 5) hydraulic design is critical to success; 6) relatively expensive; and 7) low-pressure, low-intensity lamps require a large number of lamps and acid washing to remove lamp fouling from biofilms on the exposed surfaces of the UV reactor (Metcalf and Eddy 2003). Operation requirements include: replacement of the ballasts, quartz sleeves and lamps, cleaning of fouled equipment (periodic), response to alarms, process control, and testing.

Coagulation/filtration. Maintenance issues for all chemical precipitation lagoons in cold weather regions (Canada, Michigan, and Minnesota) include: 1) seasonal algae blooms (increases TSS in effluent); 2) mixing of surface wastewater, algae and duckweed (re-suspension of precipitated solids, increase BOD and increase TSS); 3) handling, storage and mixing of chemicals; and 4) reliance on a dose chart or operator experience rather than representative sampling results. There were minimal maintenance problems associated with sludge as the accumulated amounts were consistent with buildup in the primary lagoons (Pycha and Lopez 2003)

Relative Capital Costs

The EPA (1999) wastewater technology fact sheet, *Ultraviolet Disinfection*, states that the cost of UV radiation is competitive with chlorination/dechlorination methods. The EPA (2005) wastewater technology fact sheet, *Ultraviolet Disinfection for Small Systems*, states that comparison between UV and chlorination for small systems is difficult because cost differences are based on the volume of flow.

The EPA does not include coagulation/filtration as a typical disinfection technology. Coagulation/filtration may be in place for regulations involving phosphorus removal; thus pathogen testing should be completed to determine if this methodology currently satisfies state regulations for the specific lagoon.

Constructability

Several companies provide UV radiation completely housed units that can be installed on site to reduce engineering design and construction time. Coagulation may be seasonal in the case of seasonal discharge or may be continuous. Continuous discharge may require automation, operator oversight and an end-of-line sand filter and would require additional design and construction. Note that all lagoons in Michigan with alum treatment have additional disinfection systems and a clarifier to settle the flocculent.

Recommendations

Control of pathogens is an important part of the wastewater treatment process; however, these technologies are not directly related to ammonia and nutrient removal. Most systems use chlorine for efficient pathogen removal; however this system may create undesired byproducts. The most beneficial disinfection system depends heavily on the current lagoon design. Seasonal discharge lagoons that require minimal operator interface and are not required greater than 99.9% reduction of pathogens would benefit from application of alum as a coagulant. Facultative lagoons that discharge continuously in cold weather (Montana) may best achieve pathogen reduction and limitation of adverse byproducts (from chlorine, ozone or other oxidant) with the use of UV radiation.

References

Finch, G. R. and D. W. Smith. 1986. Batch coagulation of a lagoon for fecal coliform reductions. Water Research, Vol 20, Issue 1, January 1986, pp 105-112.

H&S Environmental. 2003. Wastewater Lagoon Troubleshooting: An Operators Guide to Solving Problems and Optimizing Wastewater Lagoon Systems.

Margensin, R. and F. Schinner. 1999. Biotechnological Applications of Cold-Adapted Organisms, Springer-Verlag, Berlin, Germany.

Metcalf and Eddy. 2003. Wastewater Engineering Treatment and Reuse, Fourth Edition, McGraw Hill, New York, NY 2003

Pycha, C. and E. Lopez, 2003, Municipal Wastewater Lagoon Phosphorus Removal, Prepared by Environmental Engineers, Technical Support Section, Water Compliance Branch, U.S. EPA 5WCT-15-J, Chicago, IL, Available at: http://www.lagoonsonline.com/phosphorous.htm

United States Environmental Protection Agency (EPA). 1999. Wastewater Technology Fact Sheet, Ultraviolet Disinfection, EPA 832-R-99-064, September 1999. Available at: http://water.epa.gov/scitech/wastetech/upload/2002_06_28_mtb_uv.pdf

United States Environmental Protection Agency (EPA). 2005. Wastewater Technology Fact Sheet, Disinfection for Small Systems, 2005. Available at: http://water.epa.gov/scitech/wastetech/upload/2005_07_28_mtb_disinfection_small.pdf

4.3 Ecological Approaches

Removal of pollutants from wastewater via ecological processes occurs through complex interactions involving plants (macrophytes), microbial respiration, and fermentation. While pollutant removal is largely dependent on microbial respiration and fermentation (Faulwetter 2009), these processes are interdependent with plants. Recent studies indicate that plants play only a small direct role in nutrient removal though nutrient uptake for vegetative growth. Plant uptake generally accounts for only 3% to 19% of annual nitrogen removal and occasionally up to 25%. Total phosphorus removal via plants can be anywhere from 3% to 60% annually (Tanner 2001). Plants appear to play a more indirect role in nutrient removal by creating an environment for microbes to colonize and respire via rooting networks and the release of oxygen as well as through carbon contributions (Faulwetter et al. 2011; IWA 2000). The ability of a plant to promote certain types of microbial respiration and nutrient processing varies by species (Faulwetter et al. 2009).

4.3.1 Constructed Wetlands

General Description

Constructed wetlands are a well-established technology for water purification. The first constructed wetland in the United States was a pilot project in 1973 near Brookhaven, New York (Kadlec and Knight 1996). Constructed wetlands mimic natural wetlands which are often regarded as sinks for sediment, nutrients and debris in addition to providing habitat for wildlife, waterfowl and aquatic species. Wetlands are landscapes saturated for all or part of the year and can exhibit a variety of water tolerant plant species (Kadlec and Knight 1996). Historically wetlands have been discarded as unproductive landscapes because they are typically not usable as agricultural or rangelands unless the site is converted for land use by draining. More recent perceptions accept wetlands as important ecosystem components (Kadlec and Knight 1996).

Design criteria for treatment wetlands, usually aimed at maximizing water quality improvement, are specified in terms of substrate, sizing, hydrology (quantity, depth, and retention rates), plant selection, and future management of water and vegetation. Benefits include removal of nitrogen; both ammonium and ammonia; phosphorus; TSS; and carbon. Treatment wetlands used in conjunction with wastewater lagoons require additional space and can require imported material and/or specific native substrate material. However, these systems require less maintenance than mechanical water purification techniques.

Over the past several decades, constructed treatment wetlands designs have diversified and evolved. Three primary hydraulic designs are described in this section: vertical flow,

horizontal subsurface flow, and surface flow (Faulwetter et al. 2009; Brix 1993). See Figures 32 through 34. Each design provides a different wetland environment and redox potential, each favoring certain pollutant removal processes over others. Redox potential of the environment describes the likelihood of a transfer of an electron from a compound of a higher energy state (often organic carbon) to a compound with a lower energy state. High redox potential environments are generally found in aerobic environments. Low redox potential environments are generally found in anoxic and anaerobic environments. For more information about pollutant removal processes please refer to Section 2.0.

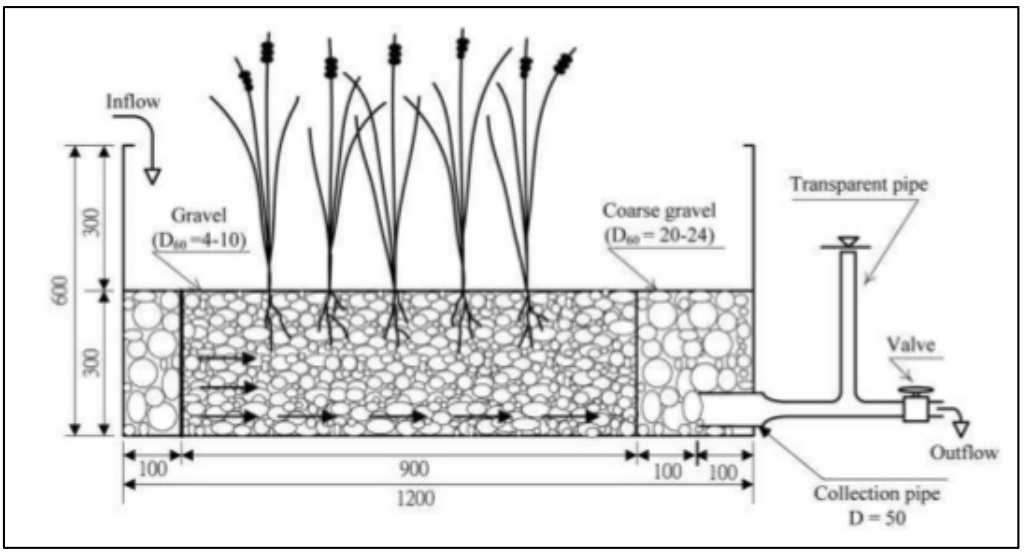

Figure 32. Subsurface flow wetland. Image: Rathor (2014).

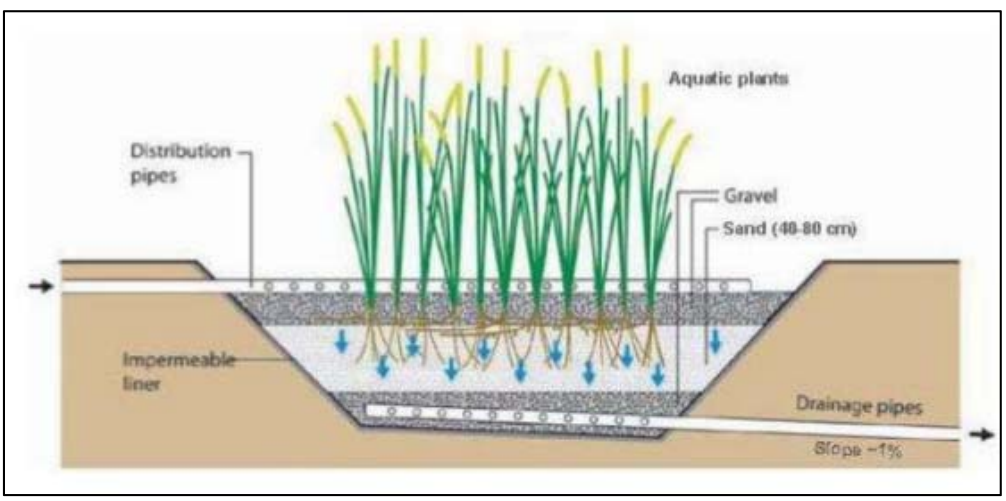

Figure 33. Vertical flow wetland. Image: Rathor (2014).

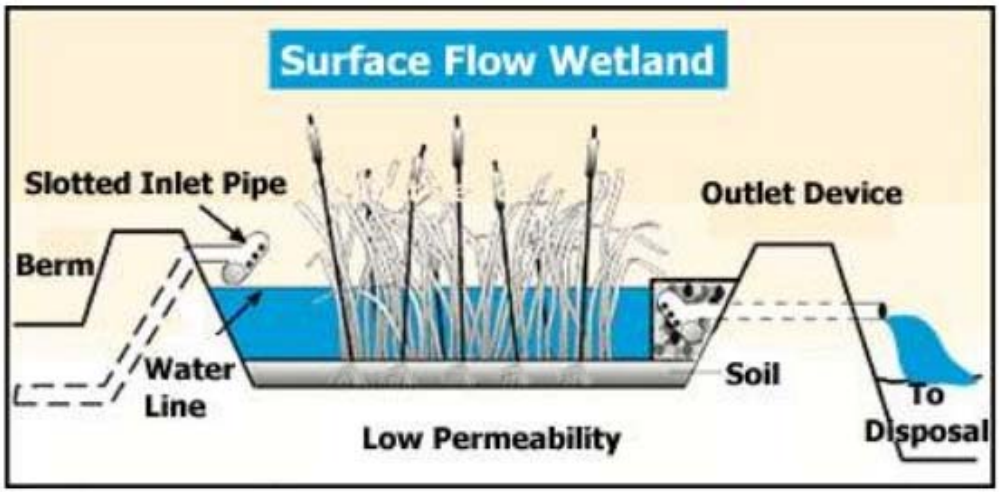

Figure 34. Vertical flow wetland. Image: Rathor (2014).

Issues Addressed

Water quality benefits and performance expectations will vary with wetland design. For example, a vertical flow wetland is more efficient at ammonium removal because of the high redox potential environment. Meanwhile, a surface flow wetland is better suited for nitrate removal because of the low redox potential environment. Phosphorus removal is largely dependent on substrate composition and thus most effective in subsurface flow wetlands where there is adequate contact time between the wastewater and substrate for phosphorus to adhere or bond with substrate particles (Garcia et al. 2010). IWA (2000) suggests that constructed wetlands are often sufficient additions to facultative lagoons to reduce TSS and BOD levels produced from algal blooms during the summer. Wetlands will filter the living and dead algae prior to releasing the effluent.

Faulwetter and others (2009) suggests that targeting specific redox potentials via design and/or management of treatment wetlands is perhaps the best method for removal of a specific pollutant or pollutants. This is because various redox potentials promote growth of a specific microbial community associated with a pollutant of concern.

An overview of pollutant removal for the three types of wetlands is provided in Tables 4, 5, and 6. Summarized data is for treatment wetlands, but not necessarily for wastewater treatment. Additionally, although concentrations are often reported for each pollutant, loading rates are not and thus surface wetlands often report high percent removal of pollutants. This may be due to lower loading rates. Descriptions of specific studies are provided below.

Table 4. Percent removal of pollutants of concern for vertical flow wetlands. Adapted from Stefanakis & Tsirintzis, 2009.

	Stefanakis and Tsirintzis 2009			Stefanakis and Tsirintzis 2009			Stefanakis and Tsirintzis 2009		
VERTICAL FLOW	**INFLUENT (mg/L)**	**EFFLUENT (mg/L)**	**PERCENT REMOVAL (%)**	**INFLUENT (mg/L)**	**EFFLUENT (mg/L)**	**PERCENT REMOVAL (%)**	**INFLUENT (mg/L)**	**EFFLUENT (mg/L)**	**PERCENT REMOVAL (%)**
Notes	Type 1*			Type 2^			Type 3+		
NO_3^-	4.7	3.4	27.7	4.7	3.4	27.7	4.7	7.8	-66.0
NO_2^-	0.6	0.9	-50	0.6	0.3	0.5	0.6	1.8	-200
NH_4^+	45.7	27.5	39.8	45.7	31.2	31.7	45.7	20.1	56.0
TKN	66.0	35.3	46.5	66.0	40.0	39.4	66.0	26.6	59.7
PO_4^{-3}	7.1	4.6	35.2	7.1	4.7	33.8	7.1L	4.1	42.3
TP	10.9	7.0	35.8	10.9	6.5	40.4	10.9	6.1	44.0
BOD	469.6	147.2	68.7	469.6	161.9	65.5	469.6	84.2	82.1
COD	568.7	194.6	65.8	568.7	217.8	61.7	568.7	124.8	78.1
TSS	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A

*Type 1 VF European wetland from the bottom consists of 15 cm cobbles, 10cm of medium sized gravel, 15cm fine gravel, 10cm sand. Only treatment W1 was used for comparison as it had the same treatment parameters as Type 2 and Type 3.
^Type 2 French; from the bottom consists of 15cm cobbles, 20cm of medium sized gravel, 50cm fine gravel.
+ Type 3 American; from the bottom consists of 15cm cobbles, 25 cm of medium sized gravel, 10cm fine gravel, 30cm sand.

Table 5. Percent removal of pollutants of concern for subsurface flow wetlands. Adapted from Kadlec & Knight, 1996, and IWA, 2000.

	IWA 2000			**IWA 2000**			**Kadlec and Knight 1996**		
SUBSURFACE FLOW	**INFLUENT (mg/L)**	**EFFLUENT (mg/L)**	**PERCENT REMOVAL (%)**	**INFLUENT (mg/L)**	**EFFLUENT (mg/L)**	**PERCENT REMOVAL (%)**	**INFLUENT (mg/L)**	**EFFLUENT (mg/L)**	**PERCENT REMOVAL (%)**
Notes	Soil based			Gravel based			Gravel based		
NO_3^-	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
NO_2^-	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
NH_4^+	N/A	N/A	N/A	N/A	N/A	N/A	5.98	4.51	**24.6**
TN	28.5	18.0	36.8	46.1	27.6	40.1	18.92	8.41	55.5
TKN	N/A	N/A	N/A	N/A	N/A	N/A	14.21	7.16	49.6
PO_4^{-3}	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
TP	8.6	6.3	26.7	6.4	3.1	51.6	4.41	2.97	32.7
BOD	97.0L	13.1	86.5	87.4	11.9	86.4	27.5	8.6	68.7
COD	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
TSS	98.6 mg/L (N=77)	13.6	86.2	71.9 (N=37)	10.8	85.0	48.2	10.3	78.6

Table 6. Percent removal of contaminants of concern in surface flow constructed wetlands. Adapted from Kadlec & Knight, 1996.

	Kadlec and Knight 1996		
SURFACE FLOW	**INFLUENT (mg/L)**	**EFFLUENT (mg/L)**	**PERCENT REMOVAL (%)**
Notes	**Gravel Based**		
NO_3^-	N/A	N/A	N/A
NO_2^-	N/A	N/A	N/A
NH_4^+	4.88	2.23	54.3
TN	9.03	4.27	52.7
TKN	7.6	4.31	43.3
PO_4^{-3}	N/A	N/A	N/A
TP	3.78	1.62	57.1
BOD	30.3	8.0	73.6
COD	N/A	N/A	N/A
TSS	45.6	13.5	70.4

Cold Weather.

Vertical Flow. In vertical flow wetlands Stefanakis and Tsihrintzis (2009) compared percent removal of pollutants above and below 15°C (wastewater temperatures did not drop below approximately 2°C). They found that nutrient removal continues at the lower temperatures, but removal of organics, BOD, COD and nitrogen decreased the most at lower temperatures. Phosphorus removal was less affected by temperature (**Table** 7). They also found that of three different designs of vertical wetlands, the design with a 30 cm sand layer atop gravels and cobbles and an 80 cm total depth was the most effective at removing pollutants both above and below 15°C.

Table 7. Percent removal of pollutants of concern for vertical flow wetlands in temperatures below 15°C. Adapted from Stefanakis and Tsirintzis 2009.

POLLUTANT	**AVERAGE PERCENT REMOVAL (%) (n=10)**
NO_3^-	N/A
NO_2^-	N/A
NH4+	39.8
TN	N/A
TKN	44.1
PO_4^{-3}	36.4
TP	34.2
BOD	65.8
COD	58.6
TSS	N/A

Subsurface and Surface Flow. In a treatment wetland exposed to temperatures below -40°C near Duluth, Minnesota, overall results indicate the wetland was able to reduce BOD, TSS, total phosphorus, total nitrogen, and FC. **Table** 8 shows these improvements.

Table 8. Pollutant concentration throughout the year in a subsurface wetland in Duluth, Minnesota. Adapted from IWA (2000).

Temperature (°C)		Winter	Spring	Summer	Fall
		2	5	15	11
TN	Influent (mg/L)	86	80	71	81
	Effluent (mg/L)	64	66	26	35
	Percent Removal (%)	**25.6**	**17.5**	**63.4**	**56.8**
TP	Influent (mg/L)	12	12	11	12
	Effluent (mg/L)	9	9	4	5
	Percent Removal (%)	**33.3**	**25**	**63.6**	**58.3**
BOD	Influent (mg/L)	251	252	176	288
	Effluent (mg/L)	34	33	14	17
	Percent Removal (%)	**86.5**	**86.9**	**92.0**	**94.1**
TSS	Influent (mg/L)	40	39	33	41
	Effluent (mg/L)	9	8	9	6
	Percent Removal (%)	**77.5**	**79.5**	**72.7**	**85.4**
FC	Influent (mg/L)	157,250	268,750	314,938	209,438
	Effluent (mg/L)	5,798	5,644	259	2,632
	Percent Removal (%)	**96.3**	**97.9**	**99.9**	**98.7**

Using data from Kadlec and Knight (1996), only performance data for those wetlands in colder climates was summarized (Table 9). Results show that surface flow wetlands are still effective at pollutant removal ranging between 50.5% and 67.6% average removal of nitrogen, phosphorus, BOD, and TSS. Only one subsurface flow wetland was documented in a colder climate. Average removal percentages are 13.5%, 45.1%, and 85% for ammonium, total nitrogen, and total phosphorus respectively.

Table 9. Pollutant removal in wetlands in colder climates. Flow volumes range between 23.3 and 4,377.8 cubic meters per day with an average of 713.4 cubic meters per day. Data record types include long-term, annual and less than annual. Adapted from Kadlec and Knight 1996.

	n	MIN PERCENT (%) REMOVAL	MAX PERCENT (%) REMOVAL	MEAN PERCENT (%) REMOVAL
SURFACE FLOW				
NH_4^+	21	-20.0	96.2	52.7
TN	12	33.5	69.5	50.5
TP	17	22.2	97.6	67.5
BOD	14	-4.0	97.7	67.6
TSS	17	-30.0	96.8	59.8

	n	MIN PERCENT (%) REMOVAL	MAX PERCENT (%) REMOVAL	MEAN PERCENT (%) REMOVAL
SUBSURFACE FLOW				
NH_4^+	1	N/A	N/A	13.5
TN	1	N/A	N/A	45.1
TP	1	N/A	N/A	85.1

Design Variations

Wetlands designs are described in detail below. Wetland environments can be further manipulated via mode of operation. Flow management options are described below under *Operations and Maintenance Issues.*

Vertical Flow Wetlands. In vertical flow wetlands wastewater is distributed across the top of the wetland and percolates vertically through the soil and/or gravel media. The soil media is composed of various layers of different substrate usually beginning with cobble at the bottom followed by coarse and then medium gravel and often topped with sand. Stefanakis and Tsihrintzis (2009) describe three different methods of substrate layering. A vertical flow wetland has a perforated pipe along the bottom of the wetland to drain filtered water. The water moving vertically through the media brings oxygen with it from the surface. This provides an aerobic environment with high redox potential which facilitates processes such as nitrification and BOD removal. Denitrification rates are low compared with the other two types of wetlands described below. Microbial activity is highest in the first 5 to 10 cm (surface) of the wetland substrate (Faulwetter et al. 2009; Brix 1993; IWA 2000).

Data compiled and summarized by Stefanakis and Tsihrintzis (2009), provides a concise report of constructed vertical wetland effectiveness. Stefanakis and Tsihrintzis (2009) compare three types of vertical flow wetlands (Type 1, 2 and 3) in an open air lab with synthetic wastewater. All were constructed of a bottom layer of cobbles, followed by medium gravels, and fine gravels. Two types had a top layer of sand. Depths of each substrate varied by vertical flow wetland type as did plantings. The authors found that the wetland topped with the thickest layer of sand (30 cm) (Type 3) was the most effective except when removing nitrates. This is despite not having the deepest overall depth of total substrate. For example, this wetland removed 56% ammonium compared to 39.8% and 31.7% removal by the Type 1 and Type 2 wetland respectively. Results are summarized in Table 4.

Subsurface Flow Wetlands. In subsurface flow wetlands, plants are installed in a gravel media. Wastewater enters at one end of a wetland, often through a filtering media such as coarse gravel, and moves horizontally though the submerged soil and plant root matrix. Waters exit the wetland at the opposite end of the wetland from which it entered. Subsurface wetlands are constructed of a gravel media to allow adequate infiltration and are often lined to prevent seepage. It is important to maintain loading rates that allow all wastewater to permeate the soil and root media. Without interaction with the soil and root media, where nutrient uptake and microbial processes occur, surface runoff will not be sufficiently treated. Subsurface wetlands are primarily

anoxic but can contain pockets of aerobic and anaerobic zones. Anoxic conditions facilitate processes such as denitrification. Redox potential is largely dependent on wetland depth, mixing and the length of the wetland. Deeper sections of the soil/rhizome matrix generally have lower redox potential unless there is vertical mixing such as upwelling. Redox potential is also generally lowest at the wetland inlet. This is where the most pollutants are located as wastewater enters the wetland and thus BOD is high. As pollutants break down, redox potential increases as wastewater approaches the outlet (Faulwetter et al. 2009; Brix 1993; IWA 2000). The potential for subsurface wetlands to vary spatially and temporally in redox potentials make these wetlands one of the more effective at wastewater treatment (Garcia et al. 2010).

IWA (2000) presents data for two horizontal flow wetlands, one constructed with a soil media, the other with a gravel media. Results show that the gravel media removed more nitrogen and phosphorus on a percent basis. Percent removal of BOD was comparable between the two media and the soil based media performed slightly better than the gravel based media for TSS. This data is shown in Table 5 above. Kadlec and Knight (1996) present summarized data from many surface flow and subsurface flow wetlands. For more detailed information including flow data, see Kadlec and Knight (1996).

Surface Flow Wetlands. In surface flow wetlands, wastewater enters the wetland at one end and moves to the other end flowing between macrophyte stems, above the soil surface. In many cases, these wetlands are not lined and wastewater can drain and filter through the soil or gravels. Surface flow wetlands are primarily anoxic. An aerobic layer may exist at the water surface from wind and other passive mixing mechanisms. Redox potential decreases with water depth, but increases with soil depth for a limited distance. This may be a result of oxygenation from roots and/or a combination of high pollutant loads settling at the soil surface (Faulwetter et al. 2009; Brix 1993). Surface flow wetlands operate best under low flow conditions, and are capable of treating relatively raw wastewater. Surface flow wetlands are no longer commonly used to treat secondary wastewater (IWA 2006). These wetlands are more appropriate for lower nutrient effluent such as storm water.

Faulwetter and others (2009) notes that although redox potential for each of these designs mentioned above can be described generally, other variables can influence redox potential such as plant species and diversity, mode of operation, and season. Additional potential variables include wetland substrate material and plant density.

Surface flow wetlands are less commonly used for wastewater treatment. In Ouray, Colorado, a surface flow wetland was added to a facultative lagoon wastewater treatment facility as a means to control TSS and BOD levels during the warmer summer months when algal blooms prevented the city from meeting regulated standards (IWA 2000).

Operations and Maintenance Issues

Operations

Faulwetter and others (2009) describes three methods of operating flows in wastewater technologies. Methods include: continuous flow feeding, batch feed, and intermittent flow feed; these methods are described in more detail below. Different methods can affect hydraulic loading rate, retention time, pollutant concentrations, and redox conditions, all which influence pollutant removal capabilities.

***Continuous flow feeding*.** Continuous flow feeding is the continuous application of wastewater to a wetland. Redox potential differs within the wetland with deeper areas exhibiting a low redox potential and shallower areas exhibiting a high redox potential. This is considered to be the least effective mode of operation when considering aerobic pollutant removal processes (Faulwetter et al. 2009).

Batch feed. Using batch feed operation, a wetland is filled with wastewater and then allowed to completely drain (via natural drainage, a pipe system, evaporation, or some other mode). The wetland is then refilled with waste water. In a controlled environment the period of time it takes to drain the water body is known and thus management actions are easily predicted. In a field setting, the period of time it takes to drain the water body may vary because of precipitation or increased influent and is not as predictable. This method of operation alternates the redox potential of the wetland or lagoon. When the wetland is filled the redox potential is low. As the wetland drains, redox potential increases. This method of operation is particularly effective for COD removal because of the oxidation period when the wetland is drained (Faulwetter et al. 2009). Stein and Hook (2005) suggest that batch feed operation may be the mode of operation more effective at pollutant removal because of the temporal variation of oxygen supply. This variation may support biofilms capable of storing nutrients for use during an alternative period of the batch cycle.

Intermittent flow feed. In intermittent flow feed, similar to batch feed, wastewater is periodically added to a wetland. Unlike batch feed, the wetland is never completely drained. This method has been shown to be more effective at ammonium removal than continuous flow feeding while maintaining the same COD removal efficiency. This method vertically mixes the wastewater adding a spatial component to the temporal variation of the batch feed mode. Both the spatial and temporal variations likely contribute to evenly distributed pollutant removal throughout the wetland (Faulwetter et al. 2009).

Maintenance

Overall maintenance for wetlands is generally low. During the first few years and particularly the first year of plant establishment, maintenance will be higher than normal. Water levels will need to be monitored and adjusted as plants grow to ensure they do not dry out or drowned and weed control is most effective during this period (Mitsch and Gosslink 2007; Tilley et al. 2014). All wetlands will require a method to control influent and effluent volumes. Options range from mechanical controls, to flash boards or simple head gate designs. From experience on previous projects, Geum

Environmental Consulting, Inc. has observed several other maintenance tasks such as maintaining water control structures, managing accumulated biomass within the wetlands, controlling beaver that may clog inflows with dams, controlling muskrats that may burrow or use wetland vegetation to make homes in wetlands, and controlling water fowl. Waterfowl use, especially in surface flow wetlands, can potentially contribute nitrogen as a result of feces. Deterring geese and other waterfowl is advised. This can be done with a network of stakes and string. This is relatively low cost; however, upkeep of the geese deterrent will add another maintenance task.

Lifespan. The longevity of all wetland types can be extended by allowing solids to settle out prior to entering the wetland (Tilley et al. 2014). The porous media used in subsurface and/or vertical wetlands may eventually be clogged with finer material and hydrologic conductivity will decrease. When this occurs, the wetland becomes a partial surface flow wetland changing in function, but still effective (Mitsch and Gosslink 2007; Tilley et al. 2014). For vertical wetlands, discontinuing use for a period of time may help unclog the porous media, allowing soils and biofilms to decompose. If full function is desired, clogging material must be removed (Tilley et al. 2014).

Risks. Mosquitoes are most problematic in surface flow wetlands where there is open water (Tilley et al. 2014). Mechanisms to avoid mosquitos are best addressed in the design phase by selecting an appropriate wetland type. Keeping BOD levels low will aid in mosquito control by providing an environment supporting mosquito prey such as aquatic insects (Kadlec and Knight 1996). The creation of swallow or bat habitat, mosquito predators, may also reduce mosquitoes. Certain plants attract mosquitoes more than others and dense stands of vegetation without open edges will reduce habitat for mosquito prey (Mitsch and Gosselink 2007; Kadlec and Knight 1996). Some species found within Montana that have a mosquito production propensity include: *Ranunculus aquatilis* and several cattail species, *Typha angustifolia*, *Typha glauca*, and *Typha latifolia*. Montana plant species that have minimal mosquito breeding problems include *Alisma trivale*, *Sagittaria latifolia*, and *Isoetes howellii*. Bacterial and chemical insecticides are an alternative for mosquito management, but many have not been fully tested and mosquitoes could develop immunity (Mitsch and Gosselink 2007). Mosquito control techniques are discussed in more detail ion Section 4.1.1.4.

Wetlands rarely exhibit odors and if so, odor is generally associated with anaerobic processes (Kadlec and Knight 1996).

Relative Capital Costs

Cost of wetland design and construction will vary with wetland design and performance expectations. Kadlec and Knight (1996) talk about construction and maintenance costs for both surface flow and subsurface flow wetlands. Because every wetland differs in design, size, and water quality goals, generalizations offer comparison among constructed wetland designs and mechanical wastewater treatment systems. Subsurface flow wetlands will be more costly than surface flow wetlands because of the need for a gravel substrate, but can potentially offer high water quality outputs. However, compared to concrete structures, either wetland will be more affordable

(Kadlec and Knight 1996). For more information regarding the cost of subsurface and surface flow wetlands see Moshiri (1993). No cost information was available for vertical wetlands. The more specific substrate criteria for vertical wetlands may require importing materials and this suggests vertical wetlands may be the most costly of all the wetlands.

Constructability

Constructed wetlands require an up-front cost for design and installation, but require minimal maintenance thereafter. An operator's license issued by Montana DEQ is required to manage wastewater lagoons and wetlands treating wastewater, but management is straightforward and practicable for small town operators. Monitoring biological parameters and adjusting flows and water levels are the main day to day tasks required to ensure the wetland is operating adequately (Kadlec and Knight 1996). Wetlands require land down slope from the lagoon, and may be more suitable on some landscapes than others, especially when considering unlined wetlands that will need to infiltrate into native soils. Construction requirements for treatment wetlands vary by wetland design (vertical, subsurface flow, or surface flow wetland) and the water quality objectives. Often a series of ponds or wetlands can be an effective means of water quality improvement (IWA 2000). Montana DEQ (2008) recommends constructing at least two wetlands providing the ability to discontinue use of one wetland for maintenance purposes. Additionally, all wetland designs should include an overflow pipe or method to control inflow volumes to prevent flooding and embankment failure.

Various references and resources are available for site specific design parameters (see Mitsch and Gosslink 2007; Kadlec and Knight 1996; Hammer and Bastian 1989; Moshiri 1993). It is advisable to involve experienced wetland experts as part of design and construction (Tilley et al. 2014). It is important to note that wetlands will require a period of establishment before optimal performance is achieved due to the time frame for plant establishment and microbial colonization (Tilley et al. 2014). Faulwetter and others (2011) suggest allowing up to three months for microbial colonies to develop. Plants will require at least one full growing season.

Sizing. One of the most important construction aspects is sizing the wetland to accommodate existing and anticipated flows. Generally, a lager wetland can treat a greater volume of influent provided wetlands are of equal depth. Various references and resources are available for site specific sizing are listed previously in this section. Kadlec and Knight (1996) provide sizing calculations for both surface flow and subsurface flow wetlands. One of the largest determinants of wetland size is the hydraulic loading rate (inflow volume/unit time/unit area); essentially how much water the wetland will receive over a period of time. For a surface flow wetland the recommended hydraulic loading rate is between 2.5 cm and 5 cm/day. For a subsurface flow wetland the recommended hydraulic loading rate is between 6 cm and 8 cm/day (Mitsch and Gosslink 2007). Detention time, the amount of time water remains in the wetland, is yet another determinant of wetland size and is recommended to be within 5 and 14 days for municipal wastewater treatment. Variable wetland depths are

beneficial in order to provide plant diversity and different redox zones promoting different nutrient cycling processes (Mitsch and Gosslink 2007).

In vertical flow wetlands, Tilley and others (2014) recommend approximately 1 to 3 square meters per person (approximately up to 0.75 acres per 1,000 persons). In colder climates up to 4 square meters per person may be required (Hoffman et al. 2011). Systems built in Norway use 7 to 9 square meters total surface area per person. Norwegian systems include both a biofilter and vertical wetland (Jenssen et al. 2005).

For subsurface flow wetlands in cold climates, it is recommended to err on the larger end of 3 to 10 square meters per person (approximately up to 2.5 acres per 1,000 persons). In a subsurface horizontal flow wetland the root depth of the selected plants represents the minimum depth to which the wetland should be designed. Within the rooting depth (or rhizosphere) sufficient oxygen should exist to support aerobic respiration (Tanner 2001).

Morphology. Vertical wetlands can be constructed below or above ground. The vertical infiltration that occurs in this type of wetland does not necessitate specified side slopes. A pipe is installed along the bottom for drainage at a recommended slope of 1% (Tilley et al. 2014).

Wetland morphology recommendations for surface and subsurface flow wetlands include slopes flatter than 6:1, 10:1 is preferable. This allows wide transition zones of varying water depth. The various water depths will support plant diversity and allow species to fill their specific niches, potentially moving into deeper or shallower waters than their designated planting zones (Mitsch and Gosslink 2007). The bottom of a subsurface wetland should slope between 0.5% and 1% to promote draining (Hoffman et al. 2011). For surface flow wetlands, Mitsch and Gosslink (2007) suggest a length to width ratio of at least 10:1 with a minimum of 2:1. This prevents short circuiting where water bypasses much of the wetland area. If necessary, more than one inflow location can be constructed in order to maximize water quality improvement by utilizing the available space. Faulwetter and others (2009) note that wetland shape can affect the redox conditions in a wetland and that considering depth and length during design can aid in obtaining the environmental conditions that support removal of pollutants of interest.

Plants. Plants develop a rooting network that provides an environment for microbial colonies to inhabit. Several studies demonstrate that including plants in a wetland, regardless of species, provides more water quality improvement than a wetland with no plants (Stefanakis and Tsihrintzis 2009; Faulwetter et al. 2009; Tanner 2001). Additionally, Faulwetter and others (2009) show that microbial communities in wetlands with vegetation are more resistant to disturbance than microbial communities in wetlands without plants. Nevertheless, too many plants may cause short circuiting and reduce retention time. This is often due to litter accumulation. Therefore, planting layouts should support even distribution of flow, and wetland management plans should include provisions for plant maintenance if needed.

Some plants have been shown to be more effective at removal of specific pollutants or perform better under certain conditions. Plant selection can depend on desired water quality results and will also depend on wetland depths. Stefanakis and Tsihrintzis (2009) found that in vertical wetlands cattails performed better than reeds at organic matter and nutrient removal. Cattails removed 71.5% BOD, 68% COD, 49% total Kjeldahl nitrogen, 42.9% ammonium, 36.5% total phosphorus, and 39.9% phosphate. Taylor and others (2009) found that in subsurface flow wetlands a variety of native plants were most successful at maintaining low COD levels throughout the year in temperatures ranging from 4°C to 24°C. Native plants included five sedges: *Carex aquatilis, Carex bebbii, Carex nebrascensis, Carex praegracilis*, and *Carex utriculata* (83-100% removal); two rushes: *Juncus arcticus* and *Juncus torreyi* (89-97% removal); *Schoenoplectus acutus*, (87-100% removal); and *Deschampsia cespitosa* (95-100% removal). Species currently more commonly used such as *Typha latifolia*, *Phragmites australis*, and *Phalaris arundinacea,* did not perform as well as the natives. COD removals only ranged between approximately 67% and 88%. Additionally, these three species have the potential to create monocultures exhibiting weedy or invasive behavior and may be undesirable for management reasons.

In colder temperatures (4°C), Taylor and others (2009) found that *Carex utriculata* and *Phalaris arundinacea* performed better than *Iris missouriensis*, *Leymus cinereus*, and *Panicum virgatum* at COD removal. Taylor and others (2009) also researched Root Oxygen Loss (ROL) in a variety of species and suggest that in colder temperatures, including plant species that supply more oxygen such as *Schoenoplectus acutus*, *Carex nebrascencis*, and *Carex utriculata* may aid in COD removal. This would also provide a higher redox potential environment encouraging microbial processes such as nitrification to reduce ammonium levels. Species with elevated redox potential during winter months include: *Juncus arcticus*, *Carex utriculata, Juncus torreyi, Schoenoplectus acutus, Carex nebrascencis, Carex praegracilis, Carex bebbii and Deschampsia cespitosa*. Research from Stein and Hook (2005) supports this work. Authors found that in cooler temperatures, redox potential was highest in wetlands planted with *Schoenoplectus acutus* followed by *Carex rostrata. Typha latifolia* showed minimal enhancement. The authors conclude that in this study, plants were a better predictor of seasonal performance variation than temperature. Within Montana, plant species selection should reflect geographic differences in soils and elevation. For example, in eastern Montana, wetland plant species are typically more tolerant of salinity than are wetland plants in western Montana.

Substrate/Matrix. Wetlands of any design are often designed with a geotextile liner to prevent infiltration into existing substrate such as with sand. The liner will help promote ponding water. Alternatively, a less porous material such as clay can be imported to act as a confining layer (Reed et al. 1988). In Montana, a liner preventing leakage of more than 6 inches per year is required to meet regulatory requirements (MT DEQ 2008)

Vertical wetlands begin with a bottom layer of gravel. Tilley and others (2014) recommend a minimum of 20 cm to allow for proper drainage. Gravels and sands are

layered above the gravel. A study by Stefanakis and Tsihrintzis (2009), described in more detail previously in this section, found that a wetland with the thicker layer of sand is most effective in removing pollutants. Stefanakis and Tsihrintzis (2009) also found that when comparing a vertical flow wetland with river bed porous media to one of carbonate material from a quarry, the river bed wetland removed pollutants to a greater degree than the carbonate media. Statistical significance was obtained for ammonium only. The river bed wetland removed 75.1% BOD, 67.6% COD, 48% total Kjeldahl nitrogen, 47.3% ammonium, and 39.3% total phosphorus. The authors also concluded that in a vertical flow wetland there is probably not enough contact time between the wastewater and media for adequate phosphorus adsorption to take place and the use of zeolite or bauxite would be better suited in a subsurface flow wetland.

Subsurface flow wetlands require slightly less substrate specifications than vertical wetlands. These wetlands need a more porous media to function such as sand or gravel, but no layering of individual materials (Mitsch and Gosslink 2007).

Surface flow wetlands require less substrate specification than the other two designs. Above the confining layer, a slightly more permeable substrate such as silt clay or loam soils are often optimal to allow root penetration and water to access plant roots (Mitsch and Gosslink 2007).

To improve phosphorus removal percentages from a surface flow or subsurface flow wetland, alum and/or iron or even phosphorus-removing chemicals can be added to the upstream end of the system or the soil media (IWA 2000). As indicated by Stefanakis and Tsihrintzis (2009), zeolite or bauxite may also be alternatives for soil amendments. See Section 2.3 for additional soil amendments that facilitate phosphorus removal.

Applicability in Montana

Slayden and Schwartz (1989) indicate that constructed wetlands work better than other alternatives in terms of reliability for wastewater treatment facilities operating in the 30,000 to 150,000 gallons per day (gpd) range; while Cueto (1993) reports that a conventional system becomes more cost effective when treating more than 5,000,000 gpd. Flows do not need to be continuous to a wetland and Mitsch and Gosslink (2007) point out that fluctuating water level is typical of natural wetlands and can improve wetland performance by supporting large plant diversity. Dry periods can also provide oxygenation periods to reduce organic loads. See *Operations and Maintenance* in this section for alternative modes of operation. Wetlands are best positioned after suspended solids have settled from the influent to limit the BOD demand allowing more oxygen for plant growth and microbial processes, especially if aerobic respiration is desired. Whether a lagoon is aerated or facultative is insignificant, although this may influence the wetland design. In Montana, DEQ recommends that effluent pass through at least two lagoon ponds prior to a surface flow wetland and primary treatment or a septic tank prior to a vertical flow wetland (MT DEQ 2008).

Recommendations

Wetlands are an effective means for treating wastewater after lagoon treatment. Wetlands require no mechanical components such as pumps and maintenance requirements are low. Overall cost is low and primarily upfront related to construction. Of the three types of wetlands reviewed, vertical wetlands generally provide an aerobic environment best suited for treating ammonium. Subsurface flow wetlands primarily provide an anoxic environment, but can also contain aerobic environments. Thus, these wetlands have the potential to treat both ammonium and nitrates. Surface wetlands can also be constructed for phosphorus removal enhancement. Surface wetlands (less commonly used for treating wastewater) primarily provide an anoxic environment best suited for treating nitrates. All wetlands will store phosphorus initially, but phosphorus removal will decline with time. Wetlands are best suited for municipalities with available land (acreage will depend on effluent volumes). Wetlands work best during the growing season and specifically in temperatures above approximately 5°C (approximately 41°F). Because of this wetlands may be best paired with other treatments or management strategies such as storage of wastewater during colder seasons or a mechanical technology that is effective in cold weather.

References

Bastian, Robert K., Peter E. Shanaghan, and Brian P. Thompson. 1989. Use of wetlands for municipal wastewater treatment and disposal – Regulatory Issue and EPA policies. In *Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural*. (ed. Donald A. Hammer), pp. 265-278. Chelsea, Michigan, USA: Lewis Publishers.

Brix, H. 1993. Wastewater treatment in constructed wetlands: system design, removal processes, and treatment performance. In *Constructed Wetlands for Water Quality Improvement* (ed. Gerald A. Moshiri), pp. 9-22. Boca Raton, FL, USA: Lewis Publishers.

Cueto, A.J. 1993. Development of criteria for the design and construction of engineered aquatic treatment units in Texas. In *Constructed Wetlands for Water Quality Improvement* (ed. Gerald A. Moshiri), pp. 99-105. Boca Raton, FL, USA: Lewis Publishers.

Dodkins, I and AF Mendzil. 2014. Enterprise Assist: Floating Treatment Wetlands (FTWs) in Water Treatment: Treatment efficiency and potential benefits of activated carbon. SEACAMS Swansea University. Prepared for FROG Environmental Ltd, Ban y Berlan, Llansadwrn, Llansadwrn, SA19 8 NA.

Faulwetter, Jennifer L., Vincent Gagnon, Carina Sundberg, Florent Chazarenc, Mark D. Burr, Jacques Brisson, Anne K. Camper, and Otto R. Stein. 2009. Microbial processes influencing performance of treatment wetlands: A review. *Ecological Engineering*, Vol. 35, pp. 987-1004.

Faulwetter, J.L., M.D. Burr, A.B. Cunningham, F.M. Stewart, A.K. Camper, and O.R. Stein. 2011. Floating treatment wetlands for domestic wastewater treatment. *Water Science & Technology*, Vol. 64, Issue 10, pp. 2089-95.

Garcia, Joan, Diederik P.L. Rousseau, Jordi Morato, Els Lesage, Victor Matamoros, and Joseph M. Bayona. 2010. Contaminant removal processes in subsurface-flow constructed wetlands. A review, *Critical Reviews in Environmental Science and Technology*, Vol. 40, Issue 7, pp. 561-661.

Hammer, Donald A. and Robert K. Bastian. 1989. Wetland ecosystems: natural water purifiers? In *Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural*. (ed. Donald A. Hammer), pp. 5-19. Chelsea, Michigan, USA: Lewis Publishers.

Hoffmann, Heike, Christoph Platzer, Martina Winker, and Elizabeth von Muench. 2011. Technology review of constructed wetlands: subsurface flow constructed wetlands for grey water and domestic wastewater treatment. Deutsche Gesellschaft fuer Internationale Zusammenarbeit (GIZ). Available at: http://www.susana.org/en/resources/library/details/930. Accessed: March 13, 2015.

IWA. 2000. Constructed wetlands for pollution control: Processes, performance, design and operation. IWA Publishing. London, UK. Available at: http://www.scribd.com/doc/208541250/Constructed-Wetlands-for-Pollution-Control-Processes-Performance-Design-and-Operation-by-IWA#scribd. Accessed March 11, 2015.

Jenssen, Peter D., Trond Maehlum, Tore Krogstad and Lasse Vrale. 2005. High performance constructed wetlands for cold climates. *Journal of Environmental Science and Health*, Vol. 40, pp. 1343-1353.

Kadlec, Robert H. and Robert L. Knight. 1996. Treatment Wetlands. CRC Press LLC. Boca Raton, Florida, U.S.A.

Lavrova, Silviya and Bogdana Koumanova. 2013. Nutrients and organic matter removal in a vertical-flow constructed wetland, applied bioremediation - active and passive approaches, Dr. Yogesh Patil (Ed.), ISBN: 978-953-51-1200-6, InTech, DOI: 10.5772/56245. Available from: http://www.intechopen.com/books/applied-bioremediation-active-and-passive-approaches/nutrients-and-organic-matter-removal-in-a-vertical-flow-constructed-wetland

Mitsch, William J. and James G. Gosslink. 2007. Wetlands. John Wiley & Sons, Inc. New Jersey, U.S.A.

Montana Department of Environmental Quality (MDEQ). 2008. Constructed wetlands for municipal wastewater treatment: Design guidelines.

Moshiri, Gerald A. 1993. Constructed Wetlands for Water Quality Improvement. CRC Press, Inc. Boca Raton, Florida, U.S.A.

Rathor, Rishabh. 2014. Constructed wetland management. Available at: http://www.slideshare.net/RishabhRathor/constructed-wetland-management. Accessed March 13, 2015.

Reed, Sherwood C., E. Joe Middlebrooks, and Ronald W. Crites. 1988. Natural Systems for Waste Management & Treatment. McGraw Hill, Inc. U.S.A.

Slayden, Robert L. Jr and Larry N. Schwartz. 1989. States' activities, attitudes and policies concerning constructed wetlands for wastewater treatment. In *Constructed Wetlands for Wastewater Treatment: Municipal, Industrial and Agricultural*. (ed. Donald A. Hammer), pp. 279-286. Chelsea, Michigan, USA: Lewis Publishers.

Stefanakis, Alexandros I. and Vassilios A. Tsihrintzis. 2009. Performance of pilot-scale vertical flow constructed wetlands treating simulated municipal wastewater: effect of various design parameters. *Desalination,* Vol. 248, pp.753-770.

Stein, Otto and Paul B. Hook. 2005. Temperature, plants and oxygen: How does season affect constructed wetland performance? Journal of Environmental Science and Health, Part A: Toxic/Hazardous Substances and Environmental Engineering. Vol. 40, pp. 1331-1342.

Tanner, C.C. 2001. Plants as ecosystem engineers in subsurface-flow treatment wetlands. *Water Science and Technology.* Vol. 44, No. 11-12, pp. 9-17.

Taylor, Carrie R., Paul B. Hook, Cathy A. Zabinski, and Otto R. Stein. 2009. Native plant material selection for water treatment wetlands. Presented at the 2009 National Meeting of the American Society of Mining and Reclamation, Billings, MT, Revitalizing the Environment: Proven Solutions and Innovative Approaches. R.I. Barnhisel (Ed.) Published by ASMR, 3134 Montavesta Rd., Lexington, KY 04502.

Tilley, Elizabeth, Lukas Ulrich, Christoph Luthi, Philippe Reymond, and Christian Zurbrugg. 2014. Compendium of sanitation systems and technologies. 2nd revised edition. Duebendorf, Switzerland: Swiss Federal Institute of Aquatic Science and Technology (Eawag). Available at: http://www.sswm.info/sites/default/files/reference_attachments/TILLEY%20et%20al%202014%20Compendium%20of%20Sanitation%20Systems%20and%20Technologies%20-%202nd%20Revised%20Edition.pdf. Accessed: March 13, 2015.

4.3.2 Vermiculture

General Description

Vermiculture is a relatively new technology where earthworms are used to aerobically treat wastewater and waste products. Vermiculture, while not yet widely implemented, is low in cost and maintenance. It is also effective at removing pollutants including: BOD, COD, TSS, and aids in nutrient cycling to make nitrogen available for plants (Bajsa et al. 2003; Sinha et al. 2008; Manyuchi et al. 2013). Vermiculture includes two processes: vermicompost and vermifiltration. In both processes, earthworms are used to purify wastewater and reduce organic loads. Vermiculture does not produce sludge. Instead the excrement from the earthworms can be used as a bio-fertilizer (Manyuchi et al. 2013). Vermiculture is applicable in colder climates.

Issues Addressed

Vermiculture can be very efficient, capturing the majority of organic materials, toxic chemicals and heavy metals. Vermiculture is especially effective at removal of nitrogen and reduction of BOD ("Biofiltro" No Date). Earthworms host nitrogen-fixing and decomposer microbes in their gut and can digest organic matter, fungi, protozoa, algae, nematodes, and bacteria (Cardoso et al. 2008; Bajsa et al. 2003; Sinha et al. 2008). In the worm gut, digestion promotes several processes that purify or aid in waste and water quality improvement. Organic matter is broken into smaller particles which create more surface area for microbes to colonize increasing microorganism populations. Larger microbial populations improve water and soil quality via microbial processes (see Section 2.0). Metals are neutralized in the worm gut further purifying the waste (Bajsa et al. 2003; Sinha et al. 2008). Additionally, earthworms convert mineralized nitrogen to ammonium which is then transformed to nitrates by microbes and is then more readily available for plant uptake (Sinha et al. 2008). In vermifiltration where wastewater surrounds the earthworms' organic wastes are also absorbed through the worm's body walls (Sinha et al. 2008). Wastewater and solids composted by earthworms can be a useful soil additive improving soil fertility (Sinha et al. 2008).

Studies. In a laboratory setting maintained at 21.5°C and 50% humidity, Sinha and others (2008) used bins containing gravel topped with garden soil to test the efficacy of earthworms at removing BOD, COD, and TSS. Earthworms were placed in a 10 cm layer of garden soil. Authors documented almost 100% removal of BOD in earthworm bins. In control bins without earthworms (only soils) percent reduction was between 70% and 85%. COD reduction in earthworm bins ranged from approximately 45% to 55% while bins without earthworms ranged from approximately 15% to 30%. TSS removal was approximately 90% in earthworm bins. In bins without earthworms percent reduction was approximately between 48% and 70%. Earthworm density was approximately 20,000 worms per cubic meter of soil. **Table** 10 summarizes these results.

Table 10. Summarize water quality parameters. Adapted from Sinha and others (2008).

Water quality parameter	Average Influent (mg/L)	Average effluent with worms (mg/L)	Average effluent without worms (mg/L)
BOD	298	3.7	68.4
COD	273.4	132.8	225
TSS	381.4	25.2	167.4

de Godoi and others (2014) reviewed several studies pertaining to vermiculture and reports adequate metals removals from both solids and wastewaters. In a laboratory setting, removal percentage using vermicompost were reported as almost 100% for copper, nickel, and zinc. Effluent was at a pH of 2.0, 7.4, and 6.8 respectively for each metal removal process. A similar study examining aqueous wastewater reported almost complete removal of cadmium, copper, chromium (as Cr_2O7^{2-}), nickel, and zinc.

Cardoso and others (2008) reviews two vermiculture facilities in rural Mexican cities. One city treated 4.8 cubic meters of sludge per month, the other 9 cubic meters per

month. Concentrations of nitrogen and phosphorus in the final product (vermicompost) were 1.35% and 0.88% for the smaller facility and 2.5% and 0.96%, respectively, for the larger facility.

Cold Weather. Vermiculture has yet to be tested in colder weather environments in the United States (personal communication Steve Harris, February 23, 2015). Biofiltro, a company based in Fresno, California and specializing in vermicompost, reports installation of facilities worldwide including the United States, New Zealand, Spain, Chile, and Brazil ("Biofiltro" No Date). A Biofiltro representative reports that a facility in Chile operates under colder conditions (personal communication, Biofiltro representative, April 13, 2015). Sinha and others (2008) reports earthworms perform best at temperatures between 20°C and 25°C, but can tolerate temperatures as low as 5°C and as high as 29°C. Sinha and others (2008) also warns that cold is less of a concern than heat. Earthworm activity is slowed with colder temperatures, but heat can cause death.

Design Variations

Both vermicompost and vermifiltration are similar to the common composting technique used in gardening that also involves earthworms. Vermicompost refers more specifically to solid waste while vermifiltration refers more specifically to wastewaters. In both scenarios, earthworms are incorporated into a composting chamber with waste or wastewater. In vermicompost, the worms are incorporated into the solid waste or sludge, which has been separated from the wastewater ("Biofiltro" No date; Bajsa et al. 2003; Sinha et al. 2008). In vermifiltration, a vermifiltration bed is constructed with layers of different sized aggregates topped with garden soil and in some cases wood chips. The garden soil provides a suitable environment for the earthworms while wastewater is pumped through the chamber and various medias ("Biofiltro" No Date; Sinha et al. 2008; Manyuchi et al. 2013). The water purification process can be as short as 4 hours ("Biofiltro" No date).

Operations and Maintenance Issues

Maintenance tasks relate to monitoring lagoon effluent and sludge or composting chamber chemistry, but generally maintenance is very low.

Lifespan. There is no documentation of lifespan in the literature reviewed. This is likely due to the lack facilities that have operated long term.

Risks. Vermicompost is marketed as an excellent soil amendment and can improve physical-chemical soil properties. However, because it is nutrient rich and can contain elevated metals, compost characteristics should be monitored to ensure soil structure is not damaged and plants are not contaminated. The most important soil characteristic to monitor is sodium. Sodium can cause microbial death (de Godoi et al. 2014). Pathogens are an additional human health concern. Solid waste is often dried which aids in pathogen reduction. However, when moisture is added (e.g. rainfall), microbes have been reported to regenerate (Bajsa et al. 2003).

Relative Capital Costs

Vermiculture is minimal in cost and maintenance compared to other technologies. Cost is primarily an upfront cost when purchasing earthworms. Earthworms will continue to reproduce at a rapid rate and purchase of additional worms is unnecessary. In Australia, the cost of 500 worms is approximately $20 as of 2008 (Sinha et al. 2008). Additional cost may be related to the installment of a composting chamber and pump to transfer waste and wastewater into the chamber. Cardoso and others (2008) cite that in a rural Mexican town a basic vermiculture facility treating 4.8 cubic meters of sludge was constructed for $10,000. Cardoso and others (2008) also states that a 1980 technical study found that vermiculture cost between $24 and $32 per ton of waste for facilities treating effluent from 50 to 500 persons. The company Biofiltro offers a multi-step vermiculture system. Biofiltro provided a cost example for a municipal wastewater treatment system treating 60,000 gallons of wastewater per day. The cost for building and operating the system was quoted at $296,000. The company clarified that depending on the analysis of the wastewater; the facility could be larger or smaller which would influence cost.

Constructability

Design Considerations. Sinha and others (2008) lists five earthworms species best suited for vermiculture: the Tiger Worm (*Eisenia fetida*), Red Tiger Worm (*E. andrei*), the Indian Blue Worm (*Perionyx excavatus*), the African Night Crawler (*Eudrilus euginae*), and the RedWorm (*Lumbricus rubellus*). These species are common in Australia. *Lumbricus terrestris* (also known as night crawlers and more common in Europe and North America) are not suggested (Bajsa et al. 2003). A combination of all or a few species is recommended. Some species may be resilient to a specific fungi and bacteria, while another may find the fungi or bacteria toxic (Bajsa et al. 2003). A variety of species will ensure a more robust earthworm community. Bajsa and others (2003) recommend monitoring the carbon to nitrogen ratio of waste as different worm species require different ratios. Generally, a carbon to nitrogen ratio of 25 is most effective at reducing soluble nitrogen and phosphates.

Hydraulic retention time and hydraulic loading rate are two of the more important design and management components to consider when implementing vermiculture practices. It is also important to maintain adequate ventilation in the soils to allow the worms to breathe (Sinha et al. 2008).

Applicability in Montana. Vermiculture can be applied to individual household composting toilets or to larger scale municipal vermicomposting (Bajsa et al. 2003). The company Biofiltro (No date), states that this technology is best suited for cities between 500 and 10,000 residents. Biofiltro offers a multi-step system that does not include lagoons. If vermiculture practices were to be used with lagoon systems in Montana, sludge and wastewater would need to be pumped from the lagoons separately and incorporated into composting chamber containing earthworms. Composting chambers differ for sludge versus wastewater treatment as described in Section 1.1 of this section. Alternatively, the vermiculture technology could potentially replace the lagoons.

Vermiculture has not been tested in the United State, however Biofiltro stated that colder climates are not a problem (personal communication, Biofiltro representative, April 13, 2015). Additionally, using earthworms to produce compost is a common gardening technique in Montana. Because vermiculture is a similar process, it seems likely this treatment is applicable to municipalities in Montana.

Recommendations

A review of vermiculture covers information from the past 17 years, but vermiculture is not a commonly used technology for wastewater purification. Water quality benefits include reduction of toxins, neutralization of metals, and nutrient processing. Maintenance is minimal and costs related to the technology are primarily up front when purchasing earthworms, constructing composting chambers, or investing in patented designs. Pumps will be required to separate sludge from wastewater. Two variations of vermiculture are referenced in the literature. Vermicompost refers specifically to solid waste and vermifiltration to and wastewaters. Vermiculture is most applicable for municipalities looking to reduce nitrogen and BOD levels with populations between 500 and 10,000 residents. Additional space is required to install pumps and composting chambers. The effectiveness of vermiculture in colder climates in the United States has not been tested. However, cold is not as large of a concern as heat. Additionally, Biofiltro reports operation of a Chile facility located in an area that receives snow. A pilot project in Montana could provide insight regarding the applicability of this technology to Montana municipalities.

References

Bajsa, O., J. Nair, K. Mathew, and G.E. Ho. 2003. Vermiculture as a tool for domestic wastewater management. *Water Science and Technology*. Vol. 28, Issue 11/12, pp. 125-132.

Biofiltro. No Date. Available at: http://biofiltro.com. Accessed April 10, 2014.

Cardoso V., Lina, Esperanza Ramirez C., and Violeta Escalante E. 2008. Vermicomposting technology for stabilizing the sewage sludge from rural waste water treatment plants. *Water Practice & Technology*, Vo. 3. No. 1. IWA Publishing. Available at: http://www.iwaponline.com/wpt/003/0003/0030003.pdf. Accessed April 10, 2015.

de Godoi Pereira, Madson, Lourdes Cardoso de Souza Neta, Maurício Paulo Ferreira Fontes, Adriana Nascimento Souza, Thaionara Carvalho Matos, Raquel de Lima Sachdev, Arnaud Victor dos Santos, Marluce Oliveira da Guarda Souza, Marta Valéria Almeida Santana de Andrade, Gabriela MarinhoMaciel Paulo, Joselito Nardy Ribeiro, and Araceli Verónica Flores Nardy Ribeiro. 2014. An overview of the environmental applicability of vermicompost: from wastewater treatment to the development of sensitive analytical methods. *The Scientific World Journal*, Vol. 2014, Article ID 917348, 14 pages.

Manyuchi, M.M., L. Kadzungura, and S. Boka. 2013. Vermifiltration of sewage wastewater for potential use in irrigation purposes using *eisenia fetida* earthworms. *World Academy of Science, Engineering and Technology*, Vol. 78, pp. 538 – 542.

Sinha, Rajiv. K., Gokul Bharmebe, and Uday Chaudhari. 2008. Sewage treatment by vermifiltration with synchronous treatment of sludge by earthworms: a low-cost sustainable technology over conventional systems with potential for decentralization. *Springer Science + Business Media, LLC*.

4.3.3 Vegetation Harvesting

General Description

Within wastewater treatments ponds, algae and duckweed (i.e., *Lemna spp., Spirodela spp*., and *Wolfiaa spp.*) have been shown to be an effective way to remove pollutants from wastewater such as coliform bacteria, COD, BOD, nitrogen, and phosphorus (Abdel-Raouf et al. 2012). Algae and duckweed have also been shown to effectively remove heavy metals in some studies (Zirchsky and Reed 1988; Abdel-Raouf et al. 2012). Algae and duckweed use nutrients such as nitrogen and phosphorus for protein synthesis (Abdel-Raouf et al. 2012). Harvesting this material removes the nutrients from the water column. If algae remain within wastewater treatment ponds there is a risk of nutrients recycling back into waters (Abdel-Raouf et al. 2012). Although costly, harvesting may increase the capacity and efficiency of algal and duckweed water treatment systems in removing of pollutants. The most common methods of algae harvesting are flocculation, coagulants, filtration, flotation, and centrifugation. Other harvesting options are both mechanical and biological. Although this technology was proposed decades ago, much of the current research is laboratory based. Additionally, cost is often prohibitive to application of this technology (Christenson 2011).

Harvesting consists of removing the algae or duckweed from the wastewater lagoon or other growing medium by concentrating the algae from a diluted suspension until it is concentrated into a paste like substance. The algae or duckweed is then dried, and processed to be used as compost, feed for livestock such as poultry, and biofuels. Macroalgae (i.e., large multicellular algae) or duckweed harvesting generally employs some level of manpower, sometimes using mechanical means. Microalgae (i.e., usually single celled algae; including cyanobacteria, diatoms, and dinoflagelattes (not true algae)), in contrast, is more often harvested by filtration, centrifugation or flocculation (Oilgae 2015). Cyanobacteria, or blue-green algae, are a type of bacteria with the ability to photosynthesize to produce energy. Diatoms are a type of phytoplankton that photosynthesize, but unlike other photosynthetic organisms their light absorbing pigments are brown not green. Dinoflagellates are single celled organisms that can be photosynthetic, but not all dinoflagellates have this ability. Dinoflagellates are a component of phytoplankton or zooplankton.

Figure 35. Lemna harvester. Image courtesy of Devils Lake (2015).

Issues Addressed

Past research and implementation of algal systems within wastewater treatment shows that algal systems remove a variety of water pollutants such as coliform bacteria, COD, BOD, nitrogen, and phosphorus (Abdel-Raouf et al. 2012; Bulent et al. 2013). Research also indicates that algal systems are capable of removing many heavy metals such as lead, cadmium, mercury, arsenic, and bromine (Abdel-Raouf et al. 2012). Two algae production technologies used to remove coliform bacteria include sewage stabilization ponds (large, shallow ponds) and high-rate stabilization ponds (designed for ideal algal growth). In most cases, high-rate stabilization ponds are better suited for algae harvesting and traditional ponds would likely not produce enough biomass to make algal harvesting a cost effective option. These technologies have reduced BOD in wastewater by 68.4% and COD by 67.2% (Colak and Kaya 1988). Other reported nitrogen reductions are between 50.2% and 86.0%. Other reported phosphorus reductions are between 85.7% and 97.8%. It is important to note that nutrients have the potential to recycle back into treated water if algae biomass is not harvested resulting in recontamination of water (Abdel-Raouf et al 2012; Bulent et al. 2013). Specific harvesting methods can also have negative effects. One example is coagulation which generally uses a coagulant that has heavy metals in its chemical makeup. This could potentially result in release of heavy metals in downstream effluent. Biomass harvested after using certain flocculants and coagulant, such as those that use metal salt, may exclude biomass from use for anything but biofuel production (Molina Grima et al. 2003; Christenson and Sims 2011).

Depending on the targeted pollutant, different algae species should be used. For wastewater treatment, *Eugelena, Oscillatoria, Chlamydomonas, Scenedesmus, Chlorella, Nitzschia, Navicula*, and *Stigeoclonium* are the most tolerant of organic pollutants (Abdel-Raouf et al. 2012). Of the various species, cyanobacteria species have faster growth rates than higher plants; a faster growth rate means higher potential for nutrient removal (Abdel-Rauof et al. 2012). Cyanobacteria can also be used to develop pigments post harvesting. In addition to type of algae chosen, other factors that affect algal growth and nutrient uptake include carbon source availability, pH, light intensity,

temperature, and biotic factors such as increased initial algal density. For example, increased initial algal density such as hyper concentrated cultures (i.e., culture where algal biomass is greater than 1.5 g/L) have been shown to increase algal growth rate and nutrient removal efficiency (Abdel-Rauof et al. 2012). In previous studies high algal concentration has been achieved by flocculation and settling using a flocculent such as chitosan. These studies have shown that ammonium and phosphorus removal rates are generally proportional to algal concentration and self-shading had little impact on results (Abdel-Rauof et al. 2012). Another biotic factor that can affect the effectiveness of algae nutrient uptake is the presence of larger vascular floating plants. These will decrease the amount of sunlight that penetrates the water which restricts the transfer of gas between water and atmosphere (IWA 2000). This interaction can result in wastewater being nearly free of algae and nearly anaerobic depending on other design parameters (IWA 2000).

Duckweed has the ability to remove carbon, nitrogen, and phosphorus from wastewater and convert it into protein along with the ability to assimilate organic materials and compounds as carbohydrates and various amino acids (Hillman 1976; Zirchsky and Reed 1988). Low nitrogen levels and high metal concentrations will inhibit duckweed biomass production (Zirschsky and Reed 1988). Hobbs (No date), reports that a *Lemna* harvesting operation, serving 10,000 residents, built in 1990 in Devils City, North Dakota removes up to 91% of wastewater total phosphorus (approximately 3.5 million gallons of wastewater per day). Typical performance values of Viet Ngo's algal system design which was used in Devils Lake is approximately 96% reduction in BOD, approximately 96% reduction in TSS, approximately 67% reduction in total phosphorus, approximately 82% reduction in total nitrogen, and approximately 93% reduction in ammonia (NETAC 1993). Duckweed can also be used to help reduce TSS by reducing algae within a wastewater lagoon. If algae become a problem, thick duckweed mats (0.5 kg/m^2 to 3.9 kg/m^2) have been shown to reduce light that infiltrates into the pond water by 34% to 94%, reducing or eliminating algae in the system (Zirchsky and Reed 1988). The resulting die off of algae from less light reduces TSS as the dead algae settle to the base of the wastewater treatment pond. If duckweed mats are thick enough, anaerobic conditions can occur. This can be beneficial because mosquito larvae cannot survive these conditions (Zirchsky and Reed 1988; Hobbs No date). In three facultative lagoons in the southeastern United States, duckweed systems have been shown to be effective in reducing BOD and TSS with an average percent reduction of 37.8% and 75.1% respectively (Zirchsky and Reed 1988). Table 11 shows duckweed uptake rates of various nutrients and heavy metals.

Table 11. Potential nutrient and heavy metals uptake rates by duckweed. Adapted from Zirchsky and Reed 1988.

Pollutant	Uptake Rate (lb/acre/yr)	Uptake Rate (lb/acre/yr)
Aluminum (Al)	260	2300
Arsenic (As)	0.6	5
Calcium (Ca)	600	5000
Chloride (Cl)	100	940

Chromium (Cr)	0.6	5
Copper (Cu)	0.1	1
Iron (Fe)	80.0	710
Magnesium (Mg)	80.0	700
Manganese (Mn)	9	80
Mercury (Hg)	0.1	1
Nitrogen (N)	611	5450
Phosphorous (P)	80	700
Potassium (K)	252	2250
Sodium (Na)	39	350
Sulfur (S)	65	580
Zinc (Zn)	0.7	6

Cold Weather. When wastewater is at about 17°C, the most commonly used duckweed genus *Lemna*, can double biomass approximately every 4 days (Hobbs No date; Zirchsky and Reed 1988). Zirchsky and Reed (1988) recommend a minimum temperature of 7°C to maintain duckweed growth. At temperatures below 7°C *Lemna* is likely to go dormant, falling to the bottom of the pond, until warm weather returns in the spring. Then, the old sunken *Lemna* fronds that survived the winter begin to photosynthesize forming air pockets causing the *Lemna* to return to the surface of the pond (Hobbs No date; Zirchsky and Reed 1988).

Design Variations

Flocculation:

A flocculent is an agent or substance that encourages suspended particles in a liquid to aggregate and form a concentrated mass that is known as a floc. Chemical flocculation is generally a pretreatment that is necessary before using other methods, such as floatation, to harvest algae (Laurent 2015; Oilgae 2015). If algae are dispersed it is difficult to separate them from the liquid medium. The flocculation process can be adjusted so that after harvesting, very few algae remain suspended within the liquid or so a small portion of the algal culture remains to recycle back to the source and use for future algae production. Determining which flocculent to use can be a process of trial and error, and is generally dependent on which algal species is used. The different makeup of each species will result in each species responding differently to various flocculants (Laurent 2015).

Flocculants are organic compounds that can be natural or synthetic. Synthetic flocculants work at pH ranges between 4.5 and 10. Many synthetic polymers currently used as flocculants are made from by-products from oil refineries; many of these chemicals or components of these chemicals are carcinogenic. This limits the use of any algae that is harvested. For most cases when flocculants have been used, algae can then only be used for biofuel production (Laurent 2015). Natural flocculants generally work between a pH of 6.0 and 7.0. Unlike synthetic flocculants they do not contain toxic monomers (Laurent 2015). Once algae is harvested it can be used in byproducts such as

compost, and some are even being used in the pharmaceutical industry (Laurent 2015). A few examples of natural flocculants are cationic starch, chitosan, and alginates (e.g. seaweed) (Molina Grima et al. 2003; Christenson and Sims 2011; Laurent 2015). Although natural flocculants are generally less effective than synthetic flocculants, some can still capture 100% of the high lipid algae during mechanical removal, and still leave enough nitrogen and phosphorus for use in the next round of biomass production within the open pond or closed loop bioreactor (Laurent 2015). For more specifics on flocculants, Molina Grima and others (2003) provide a more in depth review of various types of flocculants, their use, limitations and risks.

Coagulants:

Coagulants act like flocculants, but concentrate algae by changing soluble substances into an insoluble form. Like flocculants, coagulants are used as a pretreatment before harvesting. Currently, the most common coagulants used to harvest algae are from a class of chemicals known as alum (i.e. trivalent sulfate of metals such as aluminum, chromium, or iron and a univalent metal such as potassium or sodium) and ferric chloride (McGarry and Tonkasame 1971; Molina Grima et al. 2003; Laurent 2015). These substances are pH dependent, and many contain heavy metals (Laurent 2015). For specifics on coagulants, Molina Grima and others (2003) provides a more detailed review of their use, limitations, and risks.

Filtration:

Membranes of modified cellulose are commonly used for filtration with the aid of a suction pump (Oilgae 2015). One type of filtration, tangential filtration or cross-flow filtration, has been shown to recover 70% to 90% of algae biomass, with solids concentration of algae paste after harvesting ranging from 5% to 27% (Christenson and Sims 2011). Major limitations of this technology include membrane clogging and high costs, but it has also been shown to be reliable and results in a high solids concentration after harvesting (Christenson and Sims 2011). Another limitation is that the concentration of algae through filtration is often limited to small volumes because of clogging filters. It can also be a very slow process (Molina Grima et al. 2003; Oilgae 2015). The size of harvested algae also effects performance of the filtration process, and larger microalgae have been shown to have better removal rates than smaller microalgae species such as *Scenedesmus, Dunaliella*, and *Chlorella* (Molina Grima et al. 2003). For more specifics on filtration variations, Molina Grima and others (2003) provide a more detailed review of various methods of filtration, their use, limitations and risks.

Flotation:

Flotation is usually used in combination with flocculation or coagulation to harvest algae from wastewater, and consists of floating the algae to the surface of the wastewater or other medium where it can be removed as scum. One type of flotation, known as dissolved air floatation (DAF), uses fine air bubbles to separate algae from the wastewater and is often combined with froth floatation and flocculation (Oilgae 2015). In froth floatation, pH is adjusted to separate algae from wastewater then air bubbles are used to separate the algae allowing them to accumulate above the wastewater

where the resulting froth can be removed by suction (Oilgae 2015). Adjustments to pH for froth floatation are made depending on the species of algae used for wastewater treatment. Christenson and Sims (2011) report that with DAF, there can be 50% to 90% recovery of algae biomass, and solids concentration of algae paste after harvesting is usually 3% to 6%. Benefits of this method include that DAF has been proven effective at larger scales. However, flocculants are usually required; many of which can be toxic and will have negative downstream effects (Christenson and Sims 2011; Laurent 2015).

Centrifugation:

Christenson and Sims (2011) report that centrifugation is one of the most rapid and reliable methods for harvesting algae. In this process wastewater is placed into a centrifuge which separate liquids and solids based on density differences. One type of centrifugation is continuous-flow, which is widely used. This method is stated to be efficient, but can damage algae cells during separation (Oilgae 2015). This method is also limited because only particles with a sedimentation rate above some pre-set limiting value will be harvested (Oilgae 2015). Another type, zonal centrifugation (i.e., continuous sample-flow with isopycnic banding) has a more efficient recovery with a larger capacity than continuous-flow (Oilgae 2015). Christenson and Sims (2011) report that with centrifugation, greater than 90% of algal biomass can be recovered and solids concentration after harvesting is usually 12% to 22%. Benefits to centrifugation include reliability and a high solids concentration (Christenson and Sims 2011). Limitations include costly and energy requirements (Christenson and Sims 2011). Molina Grima and others (2003) gives a review of a few centrifugation methods that have proven to be applicable to microalgae harvesting, and Christi and Moo-Young (1991) provide additional information on selecting centrifuges for microalgae harvesting.

Mechanical:

Macroalgae, unlike microalgae, can be harvested manually using mechanical nets (Oilgae 2015). If the macroalgae is rooted in the substrate at the bottom of the pond, biomass may have to be clipped near base before using nets to collect the algae. In the past, using mechanical nets was a very labor intensive process that required removing algae from the nets one piece or clump at a time (Oilgae 2015). Some progress has been made, and now petrol driven rotary cutters can be used to help collect algae (Oligae 2015).

A few companies have designed harvesters showing promise for microalgae harvesting via mechanical means (Christenson and Sims 2011). The Shepherd Harvester from Algae to Energy, LLC uses a continuous belt that moves through an algae culture and vacuum which harvest the algae (Christenson and Sims 2011). A continuous belt harvester is also used in Algaeventure Systems, Inc.'s design which uses a primary belt to collect algae and a secondary capillary belt made of an absorbent polymer. When the secondary belt contacts the primary belt, algae is compressed to drain water before it contacts the primary belt again (Christenson and Sims 2011). These devices were not specifically designed for harvesting algal from wastewater treatment lagoons, but they likely have the ability to be retrofitted into existing lagoon systems.

In Devils Lake in North Dakota, mangers have been using duckweed biomass production in holding ponds over 58 acres to treat wastewater since 1990 (Devils Lake 2015). They used a design developed by Viet Ngo. The duckweed at this facility is harvested approximately every 3 to 5 weeks, and is dependent on duckweed density (Hobbs No date). Harvesters resemble a small tug boat, and move throughout the holding ponds skimming duckweed from the pond surface. Once harvested, the duckweed biomass is hauled off to the landfill or to farmer's fields to be used as compost (Hobbs No date).

Biological:
The Partitioned Aquaculture System (PAS) incorporates alternative techniques into conventional aquaculture ponds to increase the efficiency of removing nutrients using the patented Controlled Eutrophication Process (CEP) (Brune et al. 2007). In this system, fish ponds are separated from high-rate algal production channels in an effort to optimize excess nutrient removal by algae by increasing biomass production of algae and increasing fish yield within the fish pond. The PAS also incorporates paddlewheel mixing into the algal channels which helps maximize algal production, carbon dioxide consumption, and interfacial gas exchange rates (Brune et al. 2007). The paddlewheel also minimizes waste solids settling and allows for harvesting of excess algal solids (Brune et al. 2007). Initial results indicate separating these two aspects into different ponds can potentially increase biomass of algae 1 to 3 grams of carbon/m^2/day to 6 to 12 grams of carbon/m^2/day during the growing season (Brune et al. 2007). Effluent coming out of the algae channels then flows through fish ponds with high densities of confined filter-feeders such as tilapia where fish utilize algae as food (Brune et al. 2007; Christenson and Sims 2011). Algae and fish droppings that settle to the bottom of the aquaculture ponds are harvested by an inclined conveyer belt and fed through an anaerobic digester (Christenson and Sims 2011). The PAS aims to improve CEP nutrient removal rate by maintaining algal cell density to maximize uptake rates by filter feeders. With the PAS modification, Brune and others (2007) expect 75% to 90% of algae bound nitrogen and phosphorus to be removed without any flocculent addition. In the PAS, remaining algae will be treated with metal salt flocculate in the final polishing basin.

Understanding algae physiology has helped to create alternative methods for harvesting. Algae can be encouraged to floc without the aid of flocculants or coagulants by reducing the carbon dioxide supply within a wastewater lagoon. This process is known as "autoflocculation", and generally occurs at high pH levels (Christenson and Sims 2011; Oilgae 2015). Autoflocculation is an inconsistent process and will only occur under very specific set of conditions, which are not always repeatable. Conditions usually consist of increasing the pH which saturates the medium with calcium and phosphate ions (Christenson and Sims 2011). This can cause enough of a change in chemistry that algae concentrate. Studies that have looked at autoflocculation include: Cannell 1990; Suh and others 1997; and Lee and others 1998. Algae can also be condensed using "bioflocculation". This is when other organisms such as microbes are used to promote algal flocs (Molina Grima et al. 2003). For example, past studies have used the bacterium *Paenibacillus spp*. AM49 as a bioflocculant for harvesting *Chlorella vulgaris* (Molina Grima et al. 2003). Christenson and Sims (2011) also mention that

certain microbes have been used to flocculate algae resulting in 90% recovery. Lime can be used in a similar way. In a study conducted by Nurdogan and Oswald (1995) using lime, algae removal was above 90%.

Algae and Duckweed Biomass By-products

Compost

Duckweed can assimilate high levels of ammonium and nitrates making it a viable for use in agricultural fertilization as compost (Oron 1990).

Livestock and Poultry Feed

In order for aquatic plants to be considered good feed they should be easily harvested, low in water content, high in protein content, low in fiber and lignin content, and high in mineral absorption capabilities. They should have an extended growing season and harvesting periods, be non-toxic to both livestock and humans, be capable of being processed relatively easily, and have few pests (Culley and Epps 1973). According to Culley and Epps (1973), many duckweed species fulfill several of these criteria; one major criterium that duckweed species do not meet is having low water content. They therefore require drying post-harvest. The high protein (15% to 40%), fat (1.1% to 5.8%), nitrogen,and phosphorus (0.15% to 2.8%) content of duckweed make it a good feed source for animals such as poultry; low end of range values for protein, fat, and phosphorus are from duckweed that were grown in non-wastewater systems (Culley and Epps 1973; Zirchsky and Reed 1988; Oron 1990). Duckweed can have more protein than alfalfa, especially when grown in high nutrient wastewater (Culley and Epps 1973; Zirchsky and Reed 1988). Wastewater can contain other pollutants and contaminants other than BOD, TSS, nitrogen and phosphorus that may result in the duckweed harvest not meeting quality standards, such as toxic blue-green algae strains that may be collected along with duckweed during harvesting (Culley and Epps 1973).

Biofuel

Currently, research is being conducted to determine how efficient algae harvested from wastewater treatment systems or other biomass production facilities are as a raw material for biofuel. As of 2002, the United States EPA reported that there were more than 7,000 facultative lagoon systems in the United States, many of which could have the potential to retrofit for algal production (Christenson and Sims 2011). Some preliminary results from this research suggest algae have a large advantage over larger vascular plants for use as biofuel due to higher potential biomass production. Microalgaes grown in high rate ponds or other specialized growth reactors have the potential to produce 70 metric tons/hectare per year of ash-free dry weight compared to 3 metric tons/hectare per year from soybeans and 9 metric tons/hectare per year for corn (Abdel-Raouf et al. 2012). Using algal biomass for the production of biofuels is largely inhibited by limited cost effective harvesting methods (Christenson and Sims 2011).

Current research is looking into additives that can increase the methane production to make algae biomass a more productive biofuel. Some research has shown that adding

waste-paper to algal paste and allowing these two raw materials to anaerobically co-digest can dramatically increase methane production from less than 200 cubic centimeters of methane per liter per day to over 1600 centimeters of methane per liter per day (Brune et al. 2007). Brune and others (2007) suggest a 1:1 ratio of waste paper to algae paste.

Using wastewater lagoons can also reduce some of the costs of algae cultivation because wastewater provides large sources of nitrogen and phosphorus that are required for algae biomass production. Christenson and Sims (2011) compared the potential of various sorts of wastewater to produce algal biomass. Weak to strong domestic wastewater are estimated to be able to produce 0.3 g/L to 1.4 g/L of algal biomass respectively (Christenson and Sims 2011). Swine feedlots have the potential to produces 14.2 g/L to 37.1 g/L, and a distillery has the potential to produce 42.8 g/L (Christenson and Sims 2011).

Operations and Maintenance Issues

Longevity of algal and duckweed practice is likely most dependent on production and harvesting tools and machines rather than biological factors.

One consideration regarding maintenance of algal systems includes cleaning of filters used during filtration harvest. Another consideration regarding duckweed production and harvesting is ensuring nutrient levels in the final stage of wastewater treatment, in particular carbon, nitrogen, and phosphorus, are high enough to support vigorous duckweed biomass production. It is best to use duckweed production and harvest as a polishing stage as excessive nutrient levels or high levels of other pollutants can impede duckweed growth. A potential concern when using duckweed in wastewater lagoons is the transfer of duckweed into cells where it is not desired by birds. Duckweed primarily reproduces by asexual budding, and can double its biomass in 3 to 5 days (Hobbs no date; Zirchsky and Reed 1988). If other treatment cells need to remain duckweed free they could either be housed within an enclosed building or be covered in some way to prevent duckweed transfer. Zirchsky and Reed (1988) suggest that duckweed should be harvested approximately once weekly in systems with no algae to maintain significant nutrient removals and maintain a 20% cover of duckweed in wastewater ponds.

Risks. In algal systems, using coagulants can result in negative downstream effects from discharged effluent since most coagulants contain heavy metals. Use of coagulants also limits post-harvest options for bioproducts such as compost since heavy metal contamination may be too high to meet quality standards for these by-products. In duckweed systems with algae, authors recommend a retention time of 20 days to allow duckweed to produce mats thick enough to kill off algae and reduce BOD and TSS levels. Toxic types of blue-green algae may be in the same ponds as duckweed. One way to avoid spoiling duckweed quality is to ensure duckweed mats are thick enough to prevent algal growth. Other duckweed maintenance concerns are fungus, weather extremes, which may kill duckweed, or additional infrastructure like baffles to keep duckweeds in place on large ponds (Zirchsky and Reed 1988).

Relative Capital Costs

Currently, few technologies have proven to be cost effective to promote algal harvesting at a larger scale (i.e., greater than 90 acres) (Christenson and Sims 2011; Oilgae 2015). This is primarily because the small size of the algae in wastewater makes it difficult to collect. Unicellular eukaryotic algae are around 3 micrometers to 30 micrometers in size and cyanobacteria can be even smaller at 0.2 µm to 2µm (Christenson and Sims 2011). Molina Grima and others (2003) report that 20% to 30% of the total cost of algal biomass production is used to harvest the algae. Production costs of algae have been reported to be $0.47 to $6.93 per kg (Christenson and Sims 2011). Algal harvest as a tertiary treatment to wastewater lagoons can reduce some of the cost because nutrient levels are usually high enough that ponds do not need extra fertilization to promote algal and duckweed biomass growth.

Constructability

Production and Harvest Design. Prior to harvesting algae and/or duckweed it must first be cultivated. To produce enough biomass for large-scale harvesting, it is important to consider the following variables: nutrient supply and recycling, gas transfer and exchange, photosynthetically active radiation (PAR) delivery, algal culture integrity, environment control, and land and water availability (Christenson and Sims 2011). Generally wastewater contains sufficient nutrients to sustain production. A lack of nutrients, specifically carbon, nitrogen, and phosphorus, can require the manual addition of nutrients. Gas exchange is important because of its role in carbon production. Available light is important for photosynthetic processes that remove nutrients and sustain algal colonies. Limitation in these variables can inhibit algal and/or duckweed production. Microalgae and duckweed can be grown in fully contained photobioreactors, open ponds, and channels. Closed photobioreactors can be placed inside or outside. Most production systems, however, are placed outside to take advantage of sunlight to reduce costs of production. Design of the system is dependent on what will be harvested (i.e., microalgae, macroalgae, or duckweed), what type of harvest method will be used (i.e., flocculation, coagulation, floatation, centrifugation, mechanical, or biological), and also the desired biomass production levels. Several designs for microalgal biomass production have been proposed. Overviews of different production systems and harvest designs and for algae and duckweed can be found in: McGarry and Tonkasame (1971), Zirchsky and Reed (1988), Molina Grima and others (2003), Christenson and Sims (2011), Abdel-Raouf and others (2012), and Bulent and others (2013).

Applicability in Montana. An advantage of the filtration harvesting technique is that it allows low densities of microalgae to be collected (Oilgae 2015). However, preparation for the filtration process is generally limited to small volumes. Filtration also requires frequent unclogging of filters. Additional technologies such as a reverse-flow vacuum can prevent or prolong filter maintenance needs. This technology limits cell compaction on the filter and can allow 20 L to 30 L of wastewater with algae biomass to be filtered in 3 hours. Another technology uses a mixing blade and direct vacuum and keeps the

algae suspended in liquid preventing particles from settling during the concentration process (Oilgae 2015).
Most species of duckweed do best at pH ranges between 4.7 and 7.5. At pH levels above 10, duckweed growth is generally impeded (Zirchsky and Reed 1988). Optimum temperature ranges are 20°C to 30°C. Thick mats can produce their own heat increasing temperatures up to 10°C higher than ambient air. It is important to take this into account when temperatures rise higher than 35°C (Zirchsky and Reed 1988). Even though duckweed can tolerate higher levels of nutrients and heavy metals, excessive levels of these and polychlorinated biphenyls (PCBs) or ethylene will inhibit growth (Zirchsky and Reed 1988). When wastewater is extremely high in pollutants, a primary and secondary treatment may be needed before duckweed is used to polish wastewater. Duckweed can tolerate full sun and dense shade (Zirchsky and Reed 1988).

Post-harvest processing will depend on how the algae will be used (i.e., compost, feed, biofuel, pharmaceuticals). Harvesting usually results in a biomass paste with 5% to 15% solids (Molina Grima et al. 2003). Thus, a post-harvesting step applicable to almost all end products is drying. Drying should be done quickly after harvesting to prevent spoiling, especially in hot climates. Drying can extend the shelf-life of harvested algae. Some of the methods for drying include spray drying, drum drying, freeze-drying, and sun drying. Molina Grima and others (2003) provide a more detailed review of the mentioned drying techniques. Other post processing steps may include cell disruption and another level of filtration (Molina Grima et al. 2003).

Recommendations

Harvesting algae or duckweed is a relatively new technology but has been shown to be effective at removal of various nutrients such as phosphorus and nitrogen, and for BOD and TSS. Algae and duckweed harvest is most effective at nitrogen removal. Mechanical components are needed for harvesting vegetation. Maintenance tasks include production and harvesting of algae or duckweed. Although algae and duckweed can produce desirable products such as biofuels and compost, harvesting and converting algae and duckweed can be costly. There are several techniques for harvesting algae and duckweed. Some of the most common are flocculation, floatation, and centrifugation. Temperature and pH are two of the main environmental variables important to sustain algal and duckweed growth. Another important factor is maintaining an adequate nutrient supply that supports vigorous algae or duckweed biomass production. For wastewater treatment, algae and duckweed harvest is recommended as a secondary or tertiary treatment. Because production is low in colder climates, vegetation harvesting may be best paired with other treatments or management strategies such as storage of wastewater during colder seasons or mechanical technologies that are effective in cold weather.

References

Abdel-Raouf, N., A.A. Al-Homaidan, and I.B. M. Ibraheem. 2012. Microalgae and wastewater treatment. *Saudi Journal of Biological Sciences*, Vol. 19, pp. 257-275.

Brune, D.E., H.W. Yen, G. Schwartz, J.R. Benemann, M. J. Massingill, J. C. Van Olst, and J.R. Carlber. 2007. The controlled eutrophication process: Microalgae for biofuels

production and fertilizer recycling at the Salton Sea, California. Available at: https://www.netl.doe.gov/publications/proceedings/03/carbon-seq/PDFs/034.pdf. Accessed April 13, 2015.

Bulent, S., Tahir, M., Sonmez, F. Ali Turan Kocer, M. and O. Canpolat. 2013. Chapter 14: Relationship of algae to water pollution and wastewater treatment. Water Treatment, Dr. Walid Elshorbagy (Ed.), ISBN: 978-953-51-0928-0, InTech, DOI: 10.5772/51927. Available at: http://www.intechopen.com/books/water-treatment/relationship-of-algae-to-water-pollution-and-waste-water-treatment.

Cannell, R.J.P. 1990. Algal biotechnology. *Applied Biochemistry and Biotechnology*, Vol. 26, pp. 85-105.

Christenson, Logan. 2011. Algal biofilm production and harvesting system for wastewater treatment with biofuels by-products. *All Graduate Theses and Dissertations*. Utah State University

Christenson, L. and R. Sims. 2011. Production and harvesting of microalgae for wastewater treatment, biofuels, and bioproducts. *Biotechnology Advances*, Vol. 29, pp 686-702.

Chisti, Y., and M. Moo-Young. 1991. Fermentation technology, bioprocessing, scale-up and manufacture. In: *Biotechnology: the science and the business*. (ed. Moses M and R.E. Cape). New York: Harwood Academic Publishers, pp. 167– 209.

City of Devils Lake (Devils Lake). 2015. Devils Lake, North Dakota. Lemna Wastewater Treatment Facility. Available at: http://www.ci.devils-lake.nd.us/departments/lemna.html. Accessed April 15, 2015.

Colak, O. and Z. Kaya. 1988. A study on the possibilities of biological wastewater treatment using algae. *Turkish Journal of Biology*, Vol. 12, Issue 1, 18–19.

Culley and Epps. 1973. Use of duckweed for waste treatment and animal feed. Journal (Water Pollution Control Federation). Vol. 45, No. 2, pp. 337-347.

Gale, N.L., 1986. The role of algae and other microorganisms in metal detoxification and environmental clean-up. *Biotechnology and Bioengineering Symposium*. Vol. 16, pp. 171–180.

Hillman, W.S. 1976. Calibrating duckweeds: Light, clocks, metabolism, flowering. Science, New Series Vol. 193, Issue 4252, pp. 453-458.

Hobbs, J. No date. Tertiary treatment of wastewater by *Lemna minor* (Duckweed). Available at: http://www.dvlnd.com/documents/lemna.pdf . Accessed March 17, 2015.

Hoffman, James P. 1998. Wastewater treatment with suspended and non-suspended algae. *Journal of Phycology*, Vol. 34, pp. 757-763. Available at:

http://www.scribd.com/doc/38544231/Waste-Water-Treatment-With-Suspended#scribd. Accessed March 17, 2015.

IWA. 2000. Constructed wetlands for pollution control: Processes, performance, design and operation. IWA Publishing. London, UK. Available at: http://www.scribd.com/doc/208541250/Constructed-Wetlands-for-Pollution-Control-Processes-Performance-Design-and-Operation-by-IWA#scribd. Accessed March 11, 2015.

Laurent, E. 2015. Natural vs. synthetic flocculants. Available at: http://www.algaeindustrymagazine.com/natural-vs-synthetic-flocculents/. Accessed March 25th, 2015.

Lee, S.J., S. B. Kim, J.E. Kim, G.S. Kwon, B.D. Yoon, and H.M. Oh. 1998. Effects of harvesting method and growth stage on the flocculation of the green alga *Botryococcus braunii. Lett. Applied Microbiology,* Vol. 27, pp. 14–8.

McGarry, M.G. and C. Tonkasame. 1971. Water reclamation and algae harvesting. *Water Pollution Control Federation*, Vol. 43, Issue 5, pp. 824-835.

Molina Grima, E., E. Belarbi, F.G. Acién Fernández, A. Robles Medina, and Y. Chisti. 2003. Recovery of microalgal biomass and metabolites: process options and economics. *Biotechnology Advances*, Vol. 20, pp. 491–515.

National Environmental Technology Applications Corporation (NETAC). 1993. Environmental product profiles: Lemna System for Wastwater Treatment. Available at: http://www.dvlnd.com/documents/lemnadesr.pdf. Accessed May 6, 2015.

Nurdogan, Y. and W. J. Oswald. 1995. Enhanced nutrient removal in high-rate ponds. *Water Science Technology,* Vol. 31, pp. 33–43.

Oilgae. 2015. Algae havesting. Available at: http://www.oilgae.com/algae/har/har.html. Accessed March 15th, 2015.

Oron, G. 1990. Economic consideration in wastewater treatment with duckweed for effluent and nitrogen renovation. Journal of Water Pollution Control Federation. Vol. 62, No. 5, pp. 692-696.

Suh, H.H., G.S. Kwon, C.H. Lee, H.S. Kim, H.M. Oh, and B.D. Yoon. 1997. Characterization of bioflocculant produced by *Bacillus sp*. DP-152. *Journal of Fermentation and Bioengineering*, Vol. 84, pp. 108– 12.

Soeder, C.J., H. D. Payer, K.H. Runkel, J. Beine, and E. Briele. 1978. Sorption and concentration of toxic minerals by mass cultures of *Chlorococcales. Mitt*. *Internationale Vereinigung für Limnologie*, Vol. 21, pp. 575–584.

Zirchsky and Reed. 1988. The use of duckweed for wastewater treatment. Journal (Water Pollution Control Federation). Vol. 60, No. 7, pp.1253-1258.

4.3.4 Floating Islands

General Description

Floating islands are an established technology with potential to enhance water quality treatment in wastewater lagoons via aerobic and anaerobic processes that metabolize nutrients. Floating islands are buoyant mats that are most often planted with vegetation but can also be unvegetated. The mats float on a water surface and when planted, plant roots are exposed to the water column. These manufactured floating islands mimic natural floating islands. Natural floating islands are typically found where growth rates exceed decay rates. Buoyancy is achieved via the gasses within the vegetated mat produced through microbial metabolism of organics (Headley and Tanner 2006). The vegetated mat is composed mainly of organic matter from living and/or dead decomposed roots and rhizomes (Faulwetter et al. 2011; Batzer and Baldwin 2012).

Manufactured floating islands are available from commercial businesses, but floating islands can also be constructed independently from readily available materials. Floating islands consist of a buoyant frame and an inner matrix or simply a buoyant matrix. Floating islands can be installed in different configurations. For example, islands may be isolated and accessible only by boat. Alternatively, floating vegetation may be incorporated along a boardwalk providing access to the vegetation. When properly constructed and maintained, floating islands are durable and have withstood freeze/thaw cycles, hurricanes, major snowfalls, and tornados (Reinsel 2012a). Floating islands will thicken over time from decomposed plant material and/or other organics. (FII No Date). This thickened matrix aids buoyancy of the island by providing a space for biogas (gas produced by microbes) to accumulate. This thickening does not reduce lagoon storage volume because the matrix floats above the water and there is minimal water displacement by the below water rooting network (Reinsel 2012a).

Several water quality enhancement mechanisms are at work within floating islands, including nutrient uptake by surface vegetation (if present) and nutrient metabolism by microbes below the water surface. Microbial communities inhabit the island matrix attaching to the available surface area including the matrix itself and the rhizosphere and roots if the island is planted. Floating islands are most effective at nitrogen removal, specifically ammonium when sufficient oxygen is provided via aeration technologies such as fine or course bubble aeration (see Section 2.0 *Wastewater Treatment Processes*). Floating islands can also be effective at TSS and phosphorus removal. Using a technology that mixes the lagoon water will also improve wastewater treatment by continuously exposing the roots and microbes of the floating island to nutrient rich waters (FII No date). In addition to water quality enhancement, floating islands can provide habitat and aesthetic benefits.

Floating islands do not require additional land or earthwork and are a low maintenance water quality improvement technology. Costs for floating islands vary based on design,

size and additional technologies. Longevity is not well established, but it is estimated well-constructed and well maintained floating islands can last at least 10 years.

Figure 36. Floating Island at Rehberg Ranch, Billings, Montana. Image: (FII No date).

Issues Addressed

Various studies have shown floating islands to be effective technologies for removing nutrients from wastewater. Dodkins and Mendzil (2014) report that nutrient removal in lagoons with floating island treatments can be about 20% to 40% higher than in free surface water lagoons without floating islands. With certain management strategies this percentage can be higher. Floating islands remove nitrogen, both ammonium and nitrates, phosphorus, potassium, and TSS. Nutrient removal is achieved by plant and microbial uptake, microbial metabolism, volatilization, and sorption (Stewart et al. 2008; Dodkins and Mendzil 2014). Low TSS levels are achieved via turbulence reduction from the presence of the floating island on the water surface. The calmer waters allow suspended solids to settle (Stewart et al. 2008; Boutwell 2002; Borin 2013; Dodkins and Mendzil 2014). Borin (2013) used a TECH-AI® floating mat and reports reduction in algal growth and an increase in biofilm formations which promotes microbial colonies. After reviewing various studies evaluating floating island technologies, it is apparent floating islands are most effective at removing nitrogen, specifically ammonium, if sufficient oxygen is available. Phosphorus removal is limited and may be most effective if all plant material is routinely removed as a standard maintenance action (White and Cousins 2013). It should be noted that nutrient removal will increase as vegetation and microbial communities establish on a newly installed floating island (Faulwetter et al. 2009; Mietto et al. 2013). Faulwetter and others (2009) observed a time period of at least three months for the establishment of microbial communities.

Aeration greatly improves water quality through increased removal of ammonium via nitrification as shown by Stewart and others (2008). Aeration technologies add oxygen

to the lagoon waters. Variations and more details about this technology can be found in Section 4.2.1 *Aeration*. Studies such as Faulwetter and others (2011) examine floating islands water quality improvement under aerated conditions and do not compare results to unaerated scenarios. A study by Mietto and others (2013) concludes that nutrient removal rates were not as large as expected due to a lack of available oxygen. The authors suggest this is a result of the treatment chain layout where wastewater passes through subsurface wetlands prior to treatment by floating islands in a lagoon. A study by White and Cousins (2013) documented low DO levels in wastewater effluent which would limit nitrification. These studies demonstrate that floating islands paired with aeration technologies help maintain sufficient oxygen levels for certain nutrient cycling processes such as nitrification and may also support denitrification processes where oxygen is limited.

As described in Section 2.0 *Wastewater Treatment Processes*, a specific pollutant requires a particular set of environmental conditions to be removed from the water column. In some cases, the pollutant is removed entirely; in others it is recycled into a new compound. Dodkins and Mendzil (2014) list eight parameters that determine the water quality enhancement capabilities of floating islands: DO, carbon sources, pH, root mass, mixing, flow regimes, influent pollutant concentrations, and changes in the chemistry of the floating island over time. How these listed parameters affect nutrient processes/cycling is described in more detail in Section 2.0.

In general, floating islands remove 20% to 40% more nutrients from wastewater effluent than free water surface lagoons. Data for specific pollutants are presented in and include information collected and reviewed by Dodkins and Mendzil (2014). A subset of the studies reviewed by Dodkins and Mendzil (2014) used controls allowing a clear comparison of improved pollutant removal. In these comparable studies, total nitrogen removal increased between 12% and 42%, ammonium removal increases ranged between 6% and 48% and total phosphorus removal increased between 2% and 55% in comparison to free-surface water ponds without floating islands.

Table 12. Summary of pollutant removal of various studies including those reviewed by Dodkins and Mendzil (2014). Pollutant removals are in percent.

Study	White and Cousins 2013	Faulwetter and others 2011	Stewart and others 2008	Stewart and others 2008	Winston and others 2013^	Winston and others 2013^	Lao and others 2007 *	Lao and others 2007*	Chang and others 2013#	Van de Moortel and others 2010	Yang and others 2008
Percent Cover	95	laboratory	100	100	18	9	Unknown	Unknown	8.7	100	100
Vegetated	Yes	No	No	No	Yes	Yes	Yes	No	Unknown	Yes	Unknown
Aerated	No	Yes	Yes	No	Unknown	Unknown	Unknown	Unknown	Unknown	No	No
Notes	Non soil matrix	Plastic or shredded carpet matric	Non-woven polyester blankets	Non-woven polyester blankets	Extruded plastic matrix	Extruded plastic matrix			Storm water lagoons	Coconut coir matrix	
TN	66.9 – 87.9	91			88	48	27.2	16.4	15.8	11	31
NH_4^+			66		98	55			33.5	-2	-28
NO_3^-		73.3		100	65	82	89	69	42.6	78	91
TP	58.6 – 80.9	86.7	91		88	37	63	34	30	12	6

*Calculated based on values in graphs
^Study includes 2 different ponds both reported here.
#Values averaged over storm events and non-storm events.

Unvegetated Floating Islands. In a laboratory environment, using synthetic wastewater, Faulwetter and others (2011) reported successful COD and nitrogen removal regardless of island material. In this study, floating islands constructed of either recycled plastic or recycled carpet fibers were added to 20 cm diameter columns in a laboratory setting and compared to columns without a floating island. No floating islands were planted with vegetation. All islands were aerated and all columns operated in batch mode and continuously recirculated. Batch mode operation releases a finite volume of effluent into a pond. The pond is then allowed to drain before being refilled. The authors reported that within 2 weeks total dissolved nitrogen was reduced by 50% and then leveled off for all treatments. Ammonium and nitrates fluctuated throughout the study with controlled changes to the water chemistry. Ammonium remained in the columns after 3 to 4 weeks; however, after adding a carbon source and ceasing aeration to create an anoxic environment, denitrification increased and nitrate levels fell. When carbon was added, COD levels spiked and then rapidly returned to the previously monitored low levels within two weeks.

Stewart and others (2008) conducted a study using outdoor tanks with submerged heaters, aeration, additions of organic carbon, and covers that blocked light but allowed for gas venting. The authors found that a matrix of BioHaven® polyester blankets alone, without vegetation, are capable of creating both the aerobic environment needed to convert ammonium to nitrate and the anoxic environment needed to convert nitrate to a gaseous state. The study found that a square foot area segment of island with a thickness of 0.6 ft is capable of removing 273 mg of ammonium per day, 10,600mg of nitrate per day, and 428mg of phosphate per day per square foot unit (Stewart et al. 2008). All recorded nutrient removals assumed to be microbial as no plants existed on

these islands. The calculated removal of ammonium and phosphate was observed under aerated conditions and 27°C (80°F). Additionally, nitrate removal calculations were after the addition of a carbon source and at 27°C (80°F). These results are achieved under highly optimal conditions. Under less optimal, non-aerated conditions this study showed that a tank with a floating island removed less phosphorus than a non-aerated tank without an island.

Compared with other floating island studies cited by Stewart and others (2008), these removal rates are high, likely because of the controlled environment allowing for optimal conditions. Nitrate and ammonium removal is over twice as high and phosphate removal is almost nine times as high. All compared studies include vegetated islands and it is unknown whether these islands were aerated or not.

In a laboratory environment, with no indication of aeration Yao and others (2011) compared islands with monocultures to a control island without vegetation. Dissolved nitrogen removal rates for vegetated islands were between 0.77 mg/L and 1.87 mg/L. The island with no vegetation removed only 0.47 mg/L of dissolved nitrogen. Dissolved phosphorus rates for vegetated islands were between 2.9 mg/L and 4.1 mg/L. The island with no vegetation removed only 1.2 mg/L dissolved phosphorus. Total nitrogen was reduced by 27.2% in vegetated islands versus 16.4% in non-vegetated islands. Nitrate was reduced by 89% versus 69% in non-vegetated islands and total phosphorus was reduced by 63% versus 34% in non-vegetated islands.

A report by Dodkins and Mendzil (2014), found that although floating islands aid in nutrient removal, they also decrease DO supplies. This is optimal for denitrification and nitrate removal; however, inhibits nitrification and ammonium removal and supports the strategic use of aeration technologies.

Cold weather. In Montana, a floating island in Billings was installed in 2009 and has been successful at removing nitrogen at Rehberg Ranch. Rehberg Ranch is a small municipal system treating wastewater flows of approximately 12 gallons per minute and includes a pump for aeration and circulation. Reinsel (2012b) reports that compared to a control, the floating island removed 21% more total nitrogen. Almost all ammonium was removed and nitrate levels were kept low. The system removed 26% more ammonium than the control. Biological oxygen demand removal was almost equivalent to the control. Total suspended solid concentrations were 10% less than the control. During colder months (December through March) the floating island lagoon nutrients, total nitrogen, ammonium, total phosphorus, and phosphate, were all lower than the control. Nitrate levels were higher than the control and both the control and floating island lagoons contained nitrate volumes higher than the influent likely due to high nitrification. Total nitrogen was approximately between 6 mg/L and 17 mg/L lower than the control, peaking at about 28 mg/L. Ammonium was approximately between 4 mg/L and 17 mg/L lower than the control, peaking at about 25 mg/L. Total phosphorus was approximately between 0.4 mg/L and 2.8 mg/L lower than the control, peaking at about 7.2 mg/L. Phosphate was not sampled until March and remained a relatively consistent 1.8 mg/L lower than the control, peaking at about 6.8 mg/L. Reinsel (2012a) references

a floating island treatment in Billings Metra Park. Although this floating island treats storm water rather than wastewater, water quality results are still pertinent. Total phosphorus was reduced from 0.39 mg/L to 0.11 mg/L, and total nitrogen was reduced from 2.2 mg/L to 1.3 mg/L.

Studies such as those described above in Montana, demonstrate the ability of floating islands to function in cold weather environments. Dodkins and Mendzil (2014) point out that temperature alone does not affect phosphorus and nitrogen removal at low temperatures and that it is more a compilation of factors. However, numerous studies cite a decline in microbial processes below approximately 5°C (Johnson et al. 2005; Faulwetter et al. 2009; Dodkins and Mendzil 2014).

Design Variations

The most common floating island design is a solid, circular, island; or an island with a hollow center allowing for aeration technologies to be placed in the center of the island (FII No date; John Todd Ecological Design No date).

Another design consists of boardwalks spanning water bodies with floating vegetation on either side. This design can help direct flows, increasing retention time and pollutant removal. Aeration technology is installed between the boardwalks in the open water sections to provide sufficient oxygen, and geotextile fabrics hang from the islands to provide additional surface area for microbial colonies (Headley and Tanner 2006; John Todd Ecological Design No date).

Floating island mats are constructed with a sturdy raft or frame such as PVC pipe and an inner matrix material. The frame is often constructed or filled with buoyant material such as foam. Matrix materials range from carpet fibers, recycled plastics, gravel substrate, chicken wire, fibrous matting, coconut fiber, peat or a combination of materials (Faulwetter et al. 2011; Hubbard et al. 2004; Headley and Tanner 2006; Headley and Tanner 2011).

Studies such as Maryland Department of the Environment (2011), Vogel (2011), Boutwell (2002), and Headley and Tanner (2006) describe independently constructed floating islands. Consulting with a professional for design specifics is advisable especially in respect to sizing and plant selection.

Operations and Maintenance Issues

Floating island maintenance varies depending on design but is generally minimal as required more in the early years during plant establishment if plants are used. Those incorporating aeration or mixing will have an additional maintenance component. In some cases plant harvest may be of interest. The decay of plant material in the winter will recontribute some of the nutrients removed by plants in the spring, summer and fall (Deng and Ni 2013). It is important to keep in mind that because the majority of nutrient removal occurs through microbial processes rather than plant uptake, harvesting vegetation can be, but is not usually an effective means of enhancing nutrient removal (Dodkins and Mendzil 2014; White and Cousins 2013). Still, harvested

vegetation can be a valuable resource rich in nutrients and potential mulch material (Deng and Ni 2013). Plant harvest may actually be easier than in an established wetland because an island can be pulled to the bank, or in some instances islands are capable to holding the weight of a person (Headley and Tanner 2006).

Lifespan. Implementation of floating island technology is rather recent and does not have any long-term implementations that determine lifespan. Deng and Ni (2013) reports about a six year lifespan, while BioHaven® patented technology anticipates at least a ten year life span with some manufactures offering a five to ten year warrantee (FII No date).

Risks. During time periods where microbial activity is high and mixing limited (summer months), oxygen levels may fall so low the lagoon becomes anaerobic which can cause sulfide toxicity, killing plant roots. This most often occurs when floating islands covered more than 50% of the lagoon and will subsequently decrease the floating islands water improvement capabilities. During winter months, floating islands can increase the likelihood of icing over of lagoons because the islands calm the waters (Dodkins and Mendzil 2014).

Floating islands may require maintenance to reduce weeds and pests. For example a study by the Maryland Department of the Environment (2011) found that mallard ducks quickly inhabited the floating islands and were causing stress to the installed plants. Fences were installed to prevent duck use. Cages or netting is also suggested (Headley and Tanner 2006). Boutwell (2002) reports beaver damage to plants and the island framing and ropes. No solution has been implemented. Boutwell (2002) also reports the invasion of the common reed (*Phragmites*) and stresses the importance of minimizing such vegetation but reports no strategy for doing so. One recommendation would be to install plants with less invasive tendencies. Headley and Tanner (2006) identifies the plant establishment period as the most important time for weed management and suggests constructing islands in a way that makes them easily accessible for such activities.

Relative Capital Costs

A standard BioHaven® floating island starts at $22.50 per square foot. Additions such as walkways or mixing technologies will increase costs (personal communication, with Floating Island International, March 23, 2015).

Floating Island International has compared the cost of phosphorus removal per pound over 30 years with different treatment mechanisms. For BioHaven® islands with aeration, costs equate to $35 per pound, while a wastewater treatment plant is $220 per pound (Reinsel 2012a).

Constructability

Various floating island structures have been designed and constructed, some patented, others independent. There are few on site construction requirements for this technology other than a body of water. The use of plants enhances nutrient removal capabilities; however, floating islands without plants are shown to be effective as well.

Specific parameters effecting water quality improvement related to floating islands include floating island percent cover of the lagoon, root mass and the island matrix. Percent cover of a lagoon influences available light for photosynthesis and the aerobic/anaerobic environment. Additionally, aerobic technologies such as aeration or mixing can be added to the lagoon. Root mass is positively correlated to pollutant removal. A higher root mass within a floating islands will provide more surface area for microbes to adhere and will calm waters allowing particles to settle (Dodkins and Mendzil 2014). Below are additional construction considerations. Consultation with a professional is recommended to achieve optimal performance, but perhaps not necessary.

Design Considerations

Floating islands are installed within a body of water and thus no slope, or soil requirements apply to this technology. Islands are usually anchored to the bank in one or more locations or to the lagoon bottom to prevent the island from floating to the leeward side of the lagoon and at the same time allowing the island to rise and fall with water levels (Headley and Tanner 2011).

Island Frame and Matrix

Vogel (2011) recommends using durable, non-degradable material for the island frame to ensure longevity of the structure. An alternative to using a buoyant frame and non-buoyant inner matrix is to eliminate the frame and use a buoyant matrix (Cunningham et al. 2010).

Plants

Vogel (2011) reports that the more biomass an island begins with the better plants will establish. Native plants such as *Carex aquatilis, Carex bebbii, Carex nebrascensis, Carex praegracilis*, *Carex utriculata, Juncus arcticus, Juncus torreyi*, *Schoenoplectus acutus*, and *Deschampsia cespitosa* have been shown to be more successful at COD removal in constructed wetlands. These species are also less weedy and less likely to form monocultures than some of the commonly used species such as *Typha latifolia*, *Phramites australis*, and *Phalaris arundinacea* (Taylor et al. 2009).

Sizing

Generally floating island water quality enhancement performance is directly correlated to the floating island area (Winston et al. 2013). Sizing standards are still being developed (Headley and Tanner 2006). If using a particular patented product, professional support is available to properly size an island. Floating Island International uses a spreadsheet model to size an island to meet specific water quality objectives (FII No Date). It is important to recognize that a floating island reduces light penetration and aeration of water beneath the island. This can be desirable in some instances, but can also inhibit nitrification and other processes if cover is to too extensive. Only 20% cover is advised to achieve an aerobic environment, while 100% cover is recommended for an anaerobic environment (Dodkins and Mendzil 2014; Stewart et al. 2013; Headley and Tanner 2006).

Recommendations

Floating islands are a new but well documented technology capable of removing nutrients of concern from wastewater lagoons particularly nitrogen. Phosphorus, BOD, and TSS are also reduced. Floating islands have demonstrated a 20% to 40% increase in nutrient removal over free surface lagoons. Floating islands can be used toward the end of a lagoon system with multiple ponds and thus do not require additional land. Floating Islands require no mechanical components such as pumps and maintenance requirements are low. Overall cost is low and primarily upfront related to construction or purchase of a floating island through a distributor. Various floating island designs exist including a simple square or circular island. To improve performance, the floating island can be donut shaped and aeration can be added to the open center. Floating islands can also be constructed as boardwalks improving access to maintain the vegetation as well as open areas appropriate for aeration and methods to increase wastewater contact time with the vegetation. Floating islands work best during the growing season and specifically in temperatures above approximately 5°C (approximately 41°F). Because of this floating islands may be best paired with other treatments or management strategies such as storage of wastewater during colder seasons, mechanical technologies that are effective in cold weather, or fish (see Section 4.3.6.1).

References

Batzer, Darold P. and Andrew H. Baldwin. 2012. *Wetland Habitats of North America*. University of California Press. U.S.A.

Borin, Maurizio. 2013. Vegetated floating systems for multifunctional basins and water bodies in urban environment. Republika Slovenija Ministrstvo sa izobrazevanje, Znanost in sport. Univerza u Ljubljani.

Boutwell, John E. 2002. *Water quality and plant growth evaluations of the floating islands in Las Vegas Bay, Lake Mead, Nevada*. 2002. U.S. Department of the Interior Bureau of Reclamation. Technical Memorandum No. 8220-03-09.

Cunningham, Alfred B., Anne Camper, and Mark Burr. 2010. *Final project report: Control of microbial processes for enhanced water treatment using floating island treatment systems*. Project number 09-26. Submitted to Montana Board of Research and Commercialization Technology.

Deng, Yu and Fuquan Ni. 2013. Review of ecological floating bed restoration in polluted water. *Journal of Water Resource and Protection*, Vol. 5, pp. 1203-1209.

Dodkins, I and AF Mendzil. 2014. *Enterprise Assist: Floating Treatment Wetlands (FTWs) in water treatment: Treatment efficiency and potential benefits of activated carbon*. SEACAMS Swansea University. Prepared for FROG Environmental Ltd, Ban y Berlan, Llansadwrn, Llansadwrn, SA19 8 NA.

Faulwetter, Jennifer L., Vincent Gagnon, Carina Sundberg, Florent Chazarenc, Mark D. Burr, Jacques Brisson, Anne K. Camper, and Otto R. Stein. 2009. Microbial processes

influencing performance of treatment wetlands: A review. *Ecological Engineering*, Vol. 35, pp. 987-1004.

Faulwetter, J.L., M.D. Burr, A.B. Cunningham, F.M. Stewart, A.K. Camper and O.R. Stein. 2011. Floating treatment wetlands for domestic wastewater treatment. *Water Science & Technology*, Vol. 64, Issue 10, pp. 2089-95.

Floating Island International (FII). No date. Available at: http://www.floatingislandinternational.com. Accessed March 9, 2015.

Headley, T.R. and C.C. Tanner. 2006. Application of floating wetlands for enhanced storm water treatment: A review. Prepared for Auckland Regional Council. National Institute of Water & Atmospheric Research Ltd (NIWA). NIWA Client Report: HAM2006-123.

Headley, T.R. and C.C. Tanner. 2011. Constructed wetlands with floating emergent macrophytes: An innovative storm water treatment technology. *Critical Reviews in Environmental Science and Technology*, Vol. 42, Issue 21, pp. 2261-2310.

Hubbard, R. K., G.J. Gascho, and G.L. Newton. 2004. Use of floating vegetation to remove nutrients from Swine Lagoon wastewater. *American Society of Agricultural Engineers*, Vol. 47, Issue 6, pp. 1963-1972.

John Todd Ecological Design. No Date. Available at: http://www.toddecological.com. Accessed March 9, 2015.

Johnson, Courtney, Greg Albrecht, Quirine Ketterings, Jen Beckman, and Kristen Stockin. 2005. Nitrogen Basics – The Nitrogen Cycle. Agronomy Fact Sheet Series. Fact Sheet 2. Cornell University Cooperative Extension.

Yao Kekan, Siming Song, Zhenming Zhang, Jing Xu, Rong Zhang, Jaikai Liu, Lixia Cheng and Jinglan Lui. . 2011. Vegetation characteristics and water purification by artificial floating island. African Journal of Biotechnology. Vol. 10, No. 82, pp. 19119-19125.

Maryland Department of the Environment. 2011. Initial Assessment of the Habitat Value, Local Water Quality Impacts and Nutrient Uptake Potential of Floating Island Wetlands in the Inner Harbor, Baltimore MD.

Mietto, Anna, Maurizio Borin, Michela Salvato, Paolo Ronco, and Nicola Tadiello. 2013. Tech-IA floating system introduced in urban wastewater treatment plants in the Veneto region – Italy. *Water Science & Technology*, Vol. 68, Issue 5, pp. 1144 – 1150. Available online at: http://www.academia.edu/9303298/Tech-IA_floating_system_introduced_in_urban_wastewater_treatment_plants_in_the_Veneto_region_Italy.

Reinsel, Mark. 2012a. Floating treatment wetlands mitigate lake eutrophication. *Environmental Science & Engineering Magazin*e, May, pp. 38-41. Available at: http://www.wateronline.com/doc/floating-islands-for-stormwater-and-other-applications-0001. Accessed March 5, 2015.

Reinsel, Mark. A. 2012b. Floating Wetlands Help Boost Nitrogen Removal in Lagoons. WaterWorld. Vol. 28, Issue 6, pp. 26. Available at: http://www.waterworld.com/articles/print/volume-28/issue-6/editorial-features/floating-wetlands-help-boost--nitrogen-removal-in-lagoons.html. Accessed March 6, 2015.

Stewart, Frank M., Tim Mulholland, Alfred B. Cunningham, Bruce G. Kania and Mark T. Osterlund. 2008. Floating islands as an alternative to constructed wetlands for treatment of excess nutrients from agricultural and municipal wastes – results of laboratory-scale tests. Land Contamination and Reclamation. Vol. 16, No. 1.

Taylor, Carrie R., Paul B. Hook, Cathy A. Zabinski, and Otto R. Stein. 2009. Native plant material selection for water treatment wetlands. Presented at the 2009 National Meeting of the American Society of Mining and Reclamation, Billings, MT, Revitalizing the Environment: Proven Solutions and Innovative Approaches. R.I. Barnhisel (Ed.) Published by ASMR, 3134 Montavesta Rd., Lexington, KY 04502.

Van de Moortel, Annelies, Erik Meers, Niels De Pauw, and Filip Tack. 2010. Effects of vegetation, season and temperature on removal pollutants in experimental floating island treatment wetlands. *Water Air and Soil Pollution*, Vol. 212, Issue 1-4, pp. 281-297.

Vogel, Julie. 2011. *The Effects of Artificial Floating Wetland Island Construction Materials on Plant Biomass*. Masters of Science. Department of Environmental Science and Policy, College of Arts and Sciences. University of South Florida, St. Petersburg.

White, Sarah A. and Mathew M. Cousins. 2013. Floating treatment wetland aided remediation of nitrogen and phosphorus from simulated storm water runoff. *Ecological Engineering*, Vol. 61, pp. 207-215.

Winston, Ryan J., William F. Hunt, Shawn G. Kennedy, Laura S. Merriman, Jacob Chandler, and David Brown. 2013. Evaluation of floating treatment wetlands as retrofits to existing storm water retention ponds. *Ecological Engineering*, Vol. 54, pp. 254-265.

4.3.5 Terrestrial Application

General Description

Terrestrial application of wastewater effluent has been in practice for centuries. Application of primary effluent declined in the early 1900s as a result of outbreaks of human illness (Thomas and Harlin 1974). Secondary and tertiary effluent application is still practiced at both large and small-scale facilities. In this document, references to wastewater effluent do not include primary effluent. Wastewater effluent is most often tertiary and in some cases may be secondary. Effluent is largely used for irrigation of forage or timber harvest (City of Missoula 2013; Personal Communication, Rodger Dehaan, January 26, 2015). Irrigation of food products with effluent is subject to various regulatory conditions in Montana (ARM 2010). Another method of terrestrial application of wastewater effluent, suitable for colder climates, is conversion of effluent

to snow which is allowed to melt slowly with spring runoff (Lagoon Systems in Maine 2003a and 2003b).

Application of wastewater effluent is suitable under specific criteria relating to soil type, hydraulics, climate, and effluent volume. Generally, one of the most important criteria is a medium grained soil that allows for adequate infiltration, and that filters effluent as it moves down the soil profile (Light 1974; Thomas and Harlin 1974). Although risks of groundwater and drinking water contamination may exist, if designed and managed properly these risks are minimal and terrestrial application of wastewater can provide benefits such as nutrient rich, fertilizing waters to vegetation and economic benefits such as forage for livestock or lumber (Pescod et al. 1992; Sorber 1974). Terrestrial application of wastewater effluent is also beneficial in areas of water scarcity. The reuse of water lessens water diversion volumes from water bodies or groundwater (Pescod et al. 1992; Sorber 1974). Three methods for terrestrial application are included in this section: soil aquifer treatment allows applied wastewater effluent to percolate through the soils to groundwater, slow rate application uses wastewater effluent to irrigate vegetation, and overland flow application applies wastewater effluent to a sloped sandy or grassy area. Effluent is captured at the bottom of the slope. These various designs are described in more detail below.

Issues Addressed

Nutrient removal of soil-plant systems, such as those used in soil aquifer, slow rate and overland flow techniques, depends on soil properties, plant selection, irrigation management, and climate (Barton et al. 2005; Pescod 1992; Light 1974).

Nitrification is the primary process occurring when wastewater effluent is applied to vegetated land (Vazquez-Montiel et al. 1996). Denitrification, plant uptake, volatilization and immobilization are other mechanisms in which nitrogen can be removed from the water column. Nitrogen that does not go through one of these processes will remain in the waters and move through the soil. This process is called leaching and can allow nitrogen to reach groundwater and drinking supplies (Barton et al. 2005; Hyland et al. 2005).

Vazquez-Montiel and others (1996) discuss the potential benefits of using an effluent higher in ammonia than nitrate. Ammonia can be stored in soils while nitrate has a tendency to leach. The authors report a decrease in crop yields and increase in nitrate leaching when effluents high in nitrate were applied to vegetation. It should be noted that high concentrations of ammonia can be toxic to some plants so determining a suitable application ratio for the specific crop in use is important. Nitrate produced in the nitrification process will lower the soil pH temporarily until the vegetation takes up the nitrate.

As described in Section 2.0, phosphorus uptake mechanisms are more limited. Phosphorus can be taken up by plants or adsorbed to soil particles (Barton et al. 2005). Phosphorus is most available to plants between a pH of 6 and 7 (Hyland et al. 2005).

In **Table** 13 below, water quality parameters are summarized by application type. It should be noted that final concentrations will vary based on influent concentrations.

Table 13. Water quality concentrations of BOD, TSS, TN and TP after terrestrial application of wastewater effluent for slow rate and overland flow treatments. Adapted from Crites and others (2006).

Application	BOD (mg/L)	TSS (mg/L)	TN (mg/L)	TP (mg/L)
Slow Rate	<2	<2	<3	<0.1
Overland Flow	10	10	<10	--

*Flow equals 1 million gallons per day (mgd)

In the soil aquifer treatment BOD, TSS, and pathogens are generally effectively treated. When water quality objectives are directed toward nitrogen, management techniques can greatly influence removal rates.

In Table 14 below, water quality concentration for BOD, nitrogen, and phosphorus are summarized for the soil aquifer treatment.

Table 14. Water quality concentrations after terrestrial application of wastewater effluent for soil aquifer treatment. Adapted from Crites and others (2006).

Water quality parameter	Average applied concentration (mg/L)	Average resulting concentration (mg/L)	Percent removal (%)
BOD[a]	116.8	12.8	89.1
N[b]	26.6	9.1	65.7
P[c]	5.3	1.3	76.0

a Some examples used in the average are based on a COD rather than BOD. Average applied BOD concentrations range between 23 mg/L and 228 mg/L. Average resulting concentrations range between 1.2 mg/L and 58 mg/L.
a Average applied nitrogen concentrations range between 12 mg/L and 50 mg/L. Average resulting concentrations range between 2.8 mg/L and 19.6 mg/L.
c Final phosphorus percolate concentration are dependent on effluent travel distance. Vertical travel distances range between 2.6 feet and 60 feet. Horizontal travel distances range between 0 feet and 5580 feet when including seepage. Average applied phosphorus concentrations range between 2.1 mg/L and 10.5 mg/L. Average resulting concentrations range between 0.0 3mg/L and 1.54 mg/L.

Studies. Studies reviewed focus mainly on slow rate application and use of a profitable crop. Only one example of overland flow application was available and no studies for soil aquifer treatment were encountered. Studies are discussed below.

The city of Missoula, Montana began a small pilot project several years ago discharging disinfected wastewater effluent to 1.6 acres of hybrid poplar trees. The success of the pilot project led to a full scale project beginning in 2013. The site is now 130 acres of leased land with approximately 72,500 hybrid poplars. Approximately 10% (1,000,000 gpd) of the City's effluent is used to irrigate the area. This effluent is disinfected, meets nutrient thresholds and can legally be discharged to the Clark Fork River. The benefit in applying effluent as irrigation is gained in reducing total nutrient loads discharged to the river. Harvest of the hybrid poplars is expected in approximately 12 years and will produce wood and energy products. The City estimates that irrigation practices will reduce nitrogen discharge to the Clark Fork River by 475,000 pounds annually and phosphorus by 47,500 pounds annually (City of Missoula 2013, Chaney 2014).

The city of Victor, Montana works with a local farmer and uses wastewater effluent to irrigate a hay field. The farmer reports that the hay harvested from this field is excellent quality and that cows prefer the hay irrigated with wastewater effluent over other local sources of hay (personal communication, Rodger Dehaan, January 26, 2015).

In Portugal, Vazquez-Montiel and others (1996), compared wastewater effluent irrigation with freshwater irrigation in corn fields and found higher crop yields in the plots irrigated with wastewater effluent. Crops irrigated with wastewater effluent produced approximately 10 tons/hectare and crops irrigated with freshwater produced only 7.5 tons/hectare. Soil samples indicate that nutrient uptake of nitrate was greatest at the beginning of the growing season and declined as the crops matured. Nitrate soil levels in soils irrigated with wastewater effluent were lower than soils irrigated with freshwater until approximately 86 days into the irrigation season. Ammonia levels of wastewater irrigated soils remained lower than freshwater irrigated soils for the entire 131 day monitoring period. Phosphate levels were always higher in wastewater irrigated soils than in freshwater irrigated soils but minimally so.

Reynolds (1982) summarizes several experiments testing the efficacy of terrestrial application of wastewater. In a greenhouse experiment using secondary wastewater effluent and a corn crop, ammonium results indicate 61% was used by the corn plants, 24% was lost to denitrification and volatilization, and 14% was immobilized in the soils. In an overland flow greenhouse experiment results indicate a 66% to 80% removal of nitrogen.

Cold Weather. Both the cities of Rangely and Carrabasset Valley, Maine faced wastewater storage limitations during the winter months when infiltration was not feasible due to frozen ground. The construction of an additional lagoon was cost prohibitive leading the cities to turn to a unique alternative; the conversion of wastewater to snow for the winter months. This alternative required less space and provided a cost benefit (Lagoon Systems in Maine 2003a and 2003b).

Design Variations

Strategies for land application of wastewater include soil aquifer treatment, slow rate application and overland flow application.

In soil aquifer treatment, effluent is applied periodically to a plot of land via a sprinkler system. The land is usually not vegetated, but contains deep permeable soils. As the effluent drains through the soils, water treatment occurs through microbial processes. If effluent applications are intermittent, aerobic conditions will exist when the area dries and anoxic conditions will exist after irrigation. These two conditions allow for the removal of ammonium and nitrate respectively. The soil aquifer treatment is also an effective technology for the reduction of BOD, TSS, and pathogens in wastewater (Crites et al. 2006).

In a slow rate system, wastewater is applied to plots of land with vegetation. Infiltration rates are slower in a slow rate system than a soil aquifer treatment due to lower hydraulic loading rates. The slow rate system can include economic benefits if the

irrigated vegetation can be harvested and is of value as livestock feed or lumber. Profits from marketing these resources can help offset costs of the wastewater treatment facility. Treatment occurs largely via nutrient uptake by harvested vegetation and also by biological processes within the soils as with the soil aquifer treatment, and as described in Section 2.0 (Crites et al. 2006).

Overland Flow application of wastewater occurs on a gentle slope. Slopes are generally vegetated with low profile vegetation such as grasses. Wastewater effluent is applied to the top of the slope and the water is collected at the bottom. These systems were developed for areas with less permeable soils such as clays. Water treatment occurs via biological processes within the grass root and soil media and to a smaller extent through plant uptake. The collection of the effluent at the bottom of the system requires a discharge permit (Crites et al. 2006). This method of terrestrial application is not permitted in Montana (personal communication, Paul LeVigne, April 30, 2015).

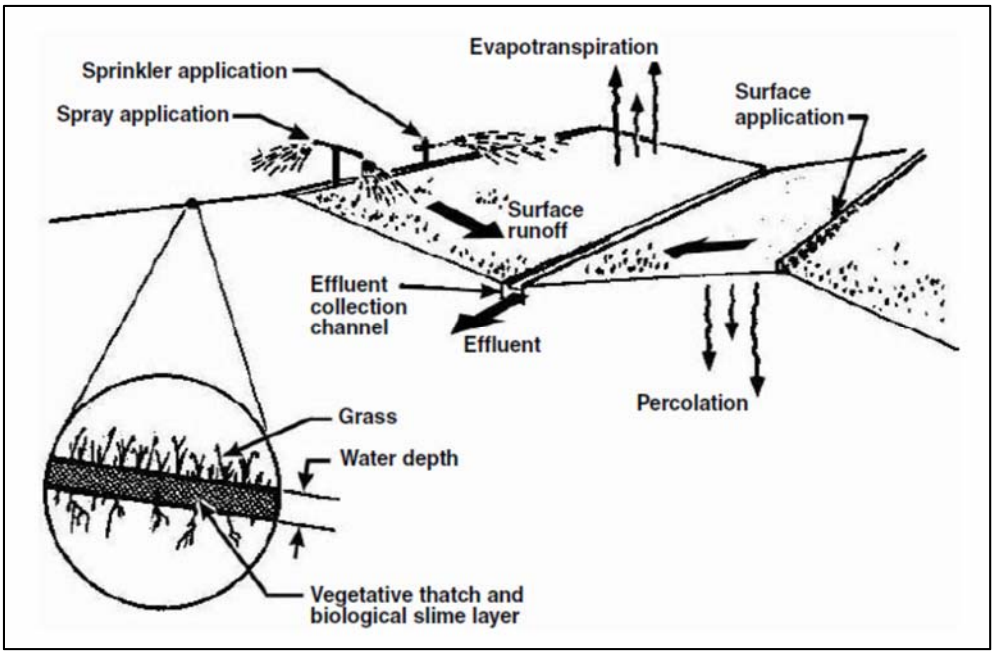

Figure 37. Overland flow schematic. From Crites et al. (2006).

For areas that experience colder weather where soils freeze and prevent infiltration of wastewater, an alternative to storing the wastewater in a constructed lagoon is converting the wastewater to snow using conventional snow makers. In this scenario, wastewater is stored on the field as snow during the winter months. As the snow melts in the spring, the waters slowly infiltrate into the soils. Two examples in Maine have found this strategy to be successful (Lagoon Systems in Maine 2003a and 2003b).

Operation and Maintenance Issues

Maintenance tasks are primarily related to harvesting vegetation in slow rate application. Vazquez-Montiel and others (1996) point out that in many cases the nutrient levels of wastewater effluent are unknown so it is often unclear what nutrients are being provided (or not provided) to plants. If applying wastewater effluent to

vegetation, it is recommended effluent is checked periodically for specific constituents such as salinity, total dissolved solids, sodium, chloride, boron, and residual chlorine. In higher quantities, these can be harmful to certain crops (Crites et al. 2006; Pescod et al. 1992). Pescod and others (1992) provides a list of agricultural species tolerant to salt and other constituents listed above.

In the soil aquifer treatment, nitrogen removal is largely dependent on management. If an operator would like to achieve an 80% or more removal of nitrogen, Crites and others (2006) suggests a loading rate of no more than 6 inches per day and a BOD to nitrogen ratio of 3:1. This provides sufficient carbon for the nitrification process. If a 3:1 ratio is not attainable the length of application should be between 7 to 9 days (generally recommendations range between 1 and 9 days) with 5 to 20 days of no application of wastewater. This allows sufficient time for microbial processes to occur. Other considerations include monitoring suspended solid loadings and BOD. High suspended solid loadings may clog surface soils and require monthly disking. High BOD loading rates may cause odors (Crites et al. 2006).

Management considerations also include irrigation methods. If concentrations of pollutants are considered too high for terrestrial application, wastewater effluent could be mixed with an alternate water source to dilute pollutant concentrations. Mixing may also create a more complex and higher quality microbial community. Another option to reduce pollutant concentrations in soils would be to alternate irrigation sources between wastewater and freshwater (Pescod et al. 1992).

Lifespan. There is no indication of lifespan in the literature; however, it seems likely wastewater application to vegetation would be similar to the lifespan of a hayfield or other agricultural field assuming pollutant concentrations do not build up to toxic levels.

Risks. Application of wastewater effluent to vegetated land may affect drinking water supplies including pathogens and even nitrogen content. Groundwater contamination is avoidable if terrestrial application is done with consideration of drinking water sources and with an understanding of local groundwater movement (Sorber 1974). EPA drinking water limits for nitrate and nitrite are 10 mg/L and 1 mg/L respectively (US EPA 2009). Effluent can be checked prior to application to vegetation, to avoid high levels of nitrogen that would contaminate groundwater.

Pescod and others (1992) note that the use of wastewater for irrigation will unlikely have negative human health effects. Concentrations of organic chemicals in municipal wastewaters are usually very low and consumption of the water would need to be excessively high and prolonged. Katz and others (2009) found only one pharmaceutical of concern, carbamazepine (an anit-convulsant) in a karstic aquifer downstream of agricultural fields sprayed with secondary wastewater effluent.

Relative Capital Costs

In Missoula, Montana where 130 acres of land is irrigated with wastewater effluent to support the harvest of hybrid poplar trees, total project cost is projected to be

$1,375,000 between 2013 and 2027. This includes the construction of the plantation (e.g. fencing, planting, etc.), management and inspection of the site, harvest, and site restoration (City of Missoula 2013).

Constructability

Construction requirements for terrestrial application of wastewater effluent are largely related to site selection including appropriate slopes, substrate and vegetation where applicable. Below are more specific site selection considerations.

Site Requirements. The amount of land required for application of wastewater effluent varies based on effluent volumes as well as nutrient concentrations. Crites and others (2006) suggests between 57 acres and 700 acres of land for slow rate application and 15 acres to 100 acres for overland flow application. Acreage is based on a design flow of 1 million gallons per day (mgd). Crites and others (2006) suggest hydraulic loading rates between 1.6 feet and 20 feet per year for slow rate application and between 10 feet and 66 feet per year for overland flow. Thomas and Harlin (1974) suggest hydraulic loading rates between 1 inch and 4 inches per week for crop irrigation and note that irrigation volumes will vary by crop. Light (1974) also suggests the use of holding tanks in case effluent application is temporarily not feasible because of weather such as excessive rain or frozen ground. Crites and others (2006) reviews appropriate slopes for various irrigation distribution systems. Slopes range from 3% for a straight furrow surface system suitable for row crops to no slope restriction for a solid set sprinkler or drip distribution system.

For a soil aquifer system, the most important site requirements are soil depth, soil permeability, depth to groundwater, and groundwater flow direction. Site soil depths and permeability will dictate hydraulic loading rates (Crites et al. 2006).

Substrate. Soil (imported or native) can influence the effectiveness of terrestrial application treatments. Soil properties to consider include physical parameters (e.g. texture, grading, liquid, and plastic limits), permeability, water-holding capacity, pH, salinity, and chemical composition (Pescod 1992). One of the most important soil properties is texture. A medium grained soil that allows for adequate infiltration and filtering of the effluent is recommended (Light 1974; Thomas and Harlin 1974). Crites and others (2006) reviews appropriate soil infiltration rates for various irrigation distribution systems. Infiltration rates range between 0.05 inches per hour to 0.3 inches per hour.

Plants. Some plants use more nutrients that others. If developing a site for terrestrial application of wastewater effluent, using species that are more effective at nutrient removal will improve the water quality of effluent water being applied as it passes through the rooting zone. Additionally, those species requiring more nutrients may benefit more from effluent application than species with lower nutrient requirements. Crites and others (2006) suggests hybrid poplars (*Populus spp*.) and Douglas-fir (*Pseudotsuga menziesii*) for western forest crops. Alfalfa is a desirable forage crop.

Table 15 below provides a list of select crops and nitrogen uptake values. Crites and others (2006) reviews appropriate crops for various irrigation distribution systems.

Table 15. Plant nitrogen uptake rates. Adapted from Crites and others (2006).

Plant	Nitrogen uptake (lb/acre/year)
Forage	
Alfalfa (*Medicago sativa*)	200-600
Brome grass (*Bromus spp*.)	115-200
Kentucky bluegrass (*Poa pratensis)*	175-240
Orchard grass (*Dactylis glomerata)*	210-250
Reed canary grass (*Phalaris arundinacea)*	300-400
Ryegrass (*Elymus spp.)*	160-250
Sweet clover (*Melilotus spp.)*	155
Tall fescue (*Festuca arundinacea)*	130-290
Western forests	
Hybrid poplar (*Populus spp*.)	270
Douglas fir (*Pseudotsuga menziesii*)	200

Applicability in Montana. To implement terrestrial application of wastewater effluent a municipality can purchase land to apply wastewater effluent if land is not already available. Alternatively, the land can be purchased and leased back to a landowner such as a farmer, or the municipality could develop a contract with a landowner. By working with a landowner the municipality can relieve some of the maintenance and management associated with crop harvest. Slow rate application requires warmer temperatures for vegetation growth and microbial activity (Crites et al. 2006). Overland flow is not permitted in Montana (personal communication, Paul LeVigne, April 30, 2015).

Recommendations

Terrestrial application of wastewater is a water purification technology that has been used for many years and is most efficient at reducing nitrogen concentrations. It can produce final effluent with concentrations lower than 2 mg/L for BOD and TSS, less than 3 mg/L for nitrogen and less than 0.1 mg/L for phosphorus. Terrestrial application requires an irrigation system, but is otherwise low tech. Maintenance is largely related to harvesting vegetation if applicable and is otherwise minimal. Overall cost is low and primarily upfront if purchasing an irrigation system or leasing land. Cost can potentially be redeemed if productive vegetation such as timber or forage is harvested. Three primary methods of terrestrial application are used: soil aquifer treatment (less common in the literature) which provides water purification via deep, filtering soils, slow rate application involving harvesting vegetation, and overland flow application which involves collecting wastewater effluent after it passes through low profile vegetation such as grass that is not harvested. Overland flow is not permitted in

Montana. Irrigating with nutrient rich wastewater effluent can produce higher crop yields and economically beneficial resources. Terrestrial application of wastewater is applicable to municipalities with access to larger plots of land or who may be able to partner with other landowners particularly farmers or ranchers. Terrestrial application of wastewater effluent can be practiced in both warmer and colder climates, although it would be necessarily seasonal in colder climates. Alternatives for terrestrial application during colder months include converting the wastewater to snow if consistently colder temperatures exist or simply storing within existing lagoons if feasible.

References

Barton, L., L.A. Schipper, G.F. Barkle, M. McLeod, T.W. Speir, M.D. Taylor, A.C. McGill, A.P. van Schaik, N.B Fitzgerald, and S.P. Pandey. 2005. Land application of domestic effluent onto four soil types: Plant uptake and nutrient leaching. *Journal of Environmental Quality*. Vol. 34, pp 635-643.

City of Missoula. 2013. Missoula Hybrid Poplar Project. Available at: http://www.nbcmontana.com/blob/view/-/20364982/data/1/-/5rsrlnz/-/Poplar-Project-Expansion-Summary.pdf. Accessed: March 22, 2015.

Chaney, Rob. 2014. 100K poplar trees to clean discharged Missoula wastewater. *Missoulian*. June 3, 2014. Available at: http://missoulian.com/news/local/k-poplar-trees-to-clean-discharged-missoula-wastewater/article_b88352b6-eaae-11e3-9588-0019bb2963f4.html. Accessed April 7, 2015.

Crites, Ronald W., Joe Middlebrooks and Sherwood C. Reed. 2006. Natural Wastewater Treatment Systems. Taylor and Francis Group, LLC. Boca Raton, Florida, U.S.A.

Hyland, Charles, Quirine Ketterings, Dale Dewing, Kristen Stockin, Karl Czmmek, Greg Albrecht, and Larry Geohring. 2005. *Agronomy Fact Sheet Series. Phosphorus Basics – The Phosphorus Cycle*. Fact Sheet 12. Cornell University Cooperative Extension. College of Agriculture and Life Science. Department of Crop and Soil Science.

Katz, Brian G., Dale W. Griffin J., and Hal Davis. 2009. Groundwater quality impacts from the land application of treatment municipal wastewater in a large karstic spring basin: Chemical and microbial indicators. *Science of the Total Environment*. Vol. 407, pp. 2872-2886.

Lagoons Systems in Maine. 2003a. Available at: http://www.lagoonsonline.com/cvsd.htm. Accessed March 25, 2015.

Lagoons Systems in Maine. 2003b. Available at: http://www.lagoonsonline.com/rangeley.htm. Accessed March 25, 2015.

Light, Ed. 1974. Land application of treated sewage in the mountain state. In *Land Application of Wastewater, Proceedings of a Research Symposium* sponsored by the U.S. Environmental Protection Agency, Region III. EPA 903-9-75-017.

Pescod, M.B. 1992. Wastewater treatment and use in agriculture – FAO irrigation and drainage paper 47. Food and Agriculture Organization of the United Nations.

Reynolds, James T. 1982. Land Application of Wastewater. Journal Water Pollution Control Federation. Vol. 54, No. 6, pp. 673-675.

Sorber, Charles A. 1974. Public health aspect of land application of wastewater effluents. In *Land Application of Wastewater, Proceedings of a Research Symposium* sponsored by the U.S. Environmental Protection Agency, Region III. EPA 903-9-75-017.

Thomas, Richard E. and Curtis C. Harlin, Jr. 1974. Land application research at Robert S. Kerr Environmental Research Laboratory. In *Land Application of Wastewater, Proceedings of a Research Symposium* sponsored by the U.S. Environmental Protection Agency, Region III. EPA 903-9-75-017.

U.S. Environmental Protection Agency (US EPA). 2009. National Primary Drinking Water Regulations. Available at: http://water.epa.gov/drink/contaminants/upload/mcl-2.pdf. Accessed: March 23, 2015.

Vazquez-Montiel, Oscar, Nigel J. Horan, and Duncan D. Mara. 1996. Management of domestic wastewater for reuse in irrigation. *Water Science and Technology*, Vol. 33, No. 10-11, pp. 355-362.

4.3.6 Other Ecological Technologies

4.3.6.1 Fish

The use of fish for wastewater treatment is referenced in other technologies such as floating islands (Section 4.3.4), vegetation harvest (Section 4.3.3), and Eco-Machines (below). The technology is not well documented. Available information is summarized below.

Fish eat various algae, microbes, detritus, fungi, protozoa, phytoplankton, zooplankton, invertebrates, and invertebrate larvae. These species use nutrients directly from wastewater (Dodkins and Mendzil 2014). When the fish are harvested, nutrients are removed from the water column. Fish harvesting can be an effective means of phosphorus removal, which is a difficult nutrient to remove using other technologies reviewed in this literature review. Harvested fish can be used as pet food, or for pork or poultry farming assuming no toxins or toxic metal contaminants are present in the wastewater and transferred to the fish. When using this technology, it is recommended to use fish adapted to lower oxygen levels (Dodkins and Mendzil 2014).

Harvesting fish from wastewater lagoons is common in South East Asia (Dodkins and Mendzil 2014). Additionally, Dodkins and Mendzil (2014) report that over 77 fish species are being used in North American treatment wetlands. In a study using floating islands, artificial aeration and fish such as silver carp (*Aristichthys nobilis)*, mandarin fish *(Siniperca chuatsi*), and crucian carp *(Carassius auratus gibelio*), approximately 34% of total nitrogen and 18% of total phosphorus was removed from the system. Of this total removal, approximately one third of removed total phosphorus and total nitrogen was removed by fish. It should be noted that fish, specifically those that feed or nest in the

sediment at the bottom of water bodies, can potentially release phosphorus from sediments (Dodkins and Mendzil 2014).

In New England, bullheads (*Ameiurus spp*) were introduced into wastewater treatment lagoons to determine if the fish would aid in sludge reduction. Results showed a sludge reduction of approximately 10 inches in the third and final lagoon cell and a decrease of sludge in the second. Reductions corresponded to the number of fish stocked in each pond. The fish were not harvested. In the spring the fish died, likely as a result of elevated ammonia concentrations common in the spring. Prior to the die off, various terrestrial animals and birds freely utilized the lagoons and fish as a food source (Lagoon Systems in Maine 2003).

Live Fuels, Inc. also utilizes fish to harvest algae. In this system fish eat algae, then those fish are harvested for oil and fishmeal. Live Fuels, Inc. does not currently apply this technique to wastewater treatment facilities, but mentions the possibility of using this technology for agricultural, industrial or municipal wastewater systems (Christenson and Sims 2011).

Recommendation

The use of fish for wastewater purification appears to be effective. Water quality benefits include a reduction in nitrogen and phosphorus when fish are harvested. Maintenance is low and relates mainly to harvesting fish. This is a viable technology for municipalities with no available land because fish can be directly introduced into existing lagoons. Pairing the use of fish with, and after, other technologies may be most beneficial to reduce high nutrient concentrations that may harm and kill fish. At the very least it may be best to use fish in final lagoons where nutrient levels are lowest. Harvesting fish could provide economically beneficial resources via the conversion of fish to pet food or compost. For lagoons in Montana, water depths would need to be deep enough (12 to 15 feet) to ensure a portion of the lagoon does not freeze over the winter providing overwintering habitat.

4.3.6.2 Eco-Machines™

An Eco-Machine™ is a patented design, by John Todd in Vermont. This design mimics natural process to reclaim water. Like a facultative lagoon, it has both aerobic and anaerobic treatment zones, but instead of being one larger pond or several smaller connected ponds like a facultative lagoon, the Eco-Machine™ isolates aerobic and anaerobic processes. Water quality improvement occurs in two main steps, a primary treatment and secondary polishing treatment. An Eco-Machine™ can be described as an artificial ecosystem of plants, microscopic algae, fungi, bacteria, protozoa, zooplankton, snails, clams, and fish working in combination within distinct treatment zones. Possible components of Eco-Machines™ are described below.

Ecological Fluidized Beds (previously known as aquatic cells), a common component of Eco-Machines™, are composed of a fluidized bed container that has a layer of course media covered by a layer of fine media. Within this container there are also

microorganisms such as benthic animals and larger aquatic plants which help with the purification of polluted water. The Ecological Fluidized Beds employ a pump to recirculate water and fine media to increase the exposure of polluted water to a community of microorganisms. Constructed wetlands, another component of Eco-Machines™, are artificial wetlands built to address water quality issues such as storm water, wastewater, or agricultural runoff. Constructed wetlands are explained in detail in Section 4.3.1 *Constructed Wetlands*. A restorer is a system of floating platforms lined with vegetation that support microbial communities on plant roots and gravel below the surface. Aeration is incorporated to optimize microbial activity (Todd 2015). A brief description of restorers can be found in Section 4.3.4 *Floating Islands*. Sediment digesters are gravel filters which sit under a layer of sediment that draw contaminated water into the greenhouse, and have an active microbial community within the gravel substrate. Mycro-reactors include a wood chip medium housing mycelium (i.e. web like tissues of fungi) used to break down contaminants as water trickles through the wood chips. Clarifiers separate microbes from treated water allowing solids to settle at the base of the clarifier; solids can be removed and composted.

Each Eco-Machine™ project combines these different components in different ways depending on site-specific needs. The majority of projects, however, include anaerobic pre-treatment components, flow equalization, and aerobic tanks as the primary treatment approach which is then followed by either Ecological Fluidized Beds or a small constructed wetland(s). Eco-Machine™ size depends on the wastewater influent volume.

An Eco-Machine™ can be placed within a greenhouse or can be a combination of aquatic cells inside a greenhouse with constructed wetlands outside of the greenhouse. In general, greenhouse based systems are recommended for areas with limited land, or in some cases in areas with colder climates. If more area is available for installation, outdoor systems with a mix of ponds and constructed wetland potentially work best and help keep energy requirements low (Todd 2015). In many cases the design incorporates multiple components such as a greenhouse, ponds, and constructed wetlands.

An Eco-Machine™ can be designed to treat a variety of nutrients and pollutants. Currently most systems are designed to treat common pollutants in wastewater such as nitrogen and phosphorus or high FC levels. However, Eco-Machine™ designs have been varied for certain projects to address contaminants such as grease and oil from food manufacturing. Eco-Machines™ have also been used to remove by products of industrial manufacturing plants from wastewater such as petroleum hydrocarbons. The Eco-Machine™ has also been used regularly to reduce BOD, TSS, and COD. Depending on the level of contamination and the project goals, effluent can be cleaned to levels that make it appropriate to produce recycled water for toilets, irrigation, or other non-potable water usage (Patsalides 2010). Available data indicates this technology is consistently effective in reducing COD, BOD, and TSS; and has potential to reduce nitrogen and phosphorus (Table 16).

Table 16. Percent reduction of nutrient and pollutants of influent after treatment through an Eco-Machine™. Data is compiled from Todd (2015).

Case Study	COD	BOD	TSS	TN	TKN	NH4	N03	TP
Fisherville Canal near Grafton, Massachusetts	--	--	--		--	--	--	--
Baima Canal Fuzhou, China	92	92						
City of South Burlington, Vermont	93	97.3 (summer)	97.2	90.4	94.3	98.2		54.1
Ethel M. Chocolates Henderson, Nevada		99.5	98.4					
Tyson Corporation Berlin Maryland	95	96 (summer) 92 (winter)	95				35	

Maintenance requirements for Eco-Machines™ are minimal, and may be concentrated within the first year or two of operation while operational strategies are developed and plants establish. The incorporation of various treatment techniques makes Eco-Machines™ a flexible technology for both water treatment goals and administrative constraints. Eco-Machines™ are generally a more expensive and elaborate option for wastewater treatment for smaller municipalities in Montana, though where applicable; this technology could be an attractive educational tool for the public in addition to a wastewater treatment facility.

Recommendation

The incorporation of various treatment techniques makes Eco-Machines™ a flexible technology for both water treatment goals and administrative constraints. Targeted pollutants will vary with the treatment techniques selected. Maintenance requirements for Eco-Machines™ are minimal, and may be concentrated within the first year or two of operation while operational strategies develop and plants establish. Eco Machines™ are generally a more expensive and elaborate option for wastewater treatment for smaller municipalities in Montana; although, where applicable, this technology could be an attractive educational tool for the public in addition to a wastewater treatment facility. A variety of components included in Eco-Machines™ perform best during the growing season. Additional components, specifically greenhouses, can ensure year ground performance of wastewater treatment.

References

Boynton, D. 2010. $671,000 grant for Eco-Machine™ targets Fisherville Mill site water. News Telegram.com. Available at: http://www.telegram.com/article/20100806/NEWS/8060459/1101. Accessed March 21, 2015.

Christenson, L. and R. Sims. 2011. Production and harvesting of microalgae for wastewater treatment, biofuels, and bioproducts. *Biotechnology Advances*, Vol. 29, pp 686-702.

Dodkins, I and AF Mendzil. 2014. Enterprise Assist: Floating Treatment Wetlands (FTWs) in water treatment: Treatment efficiency and potential benefits of activated carbon. SEACAMS Swansea University. Prepared for FROG Environmental Ltd, Ban y Berlan, Llansadwrn, Llansadwrn, SA19 8 NA.

Lagoon Systems in Maine. 2003. Utilizing the bullhead for sludge management. Available at: http://www.lagoonsonline.com/hornpout.htm. Accessed April 10, 2015.

O'brien, L. 2011. Green wastewater treatment systems don't work as well as hoped. Daily Journal of Commerce Oregon. Available at: http://djcoregon.com/news/2011/11/09/green-wastewater-treatment-systems-dont-work-as-well-as-hoped/#ixzz3VQpIwIEE. Accessed March 21st, 2015.

Patsalides, L. 2010. Wastewater treatment: A closer look at Eco Machine's™ biomimicry solutions. Bright Hub. Available at: http://www.brighthub.com/environment/science-environmental/articles/50029.aspx. Accessed March 21st, 2015.

Todd, J. 2015. Eco-Machines. John Todd Ecological Design. Available at: http://toddecological.com/eco-machines. Accessed March 24, 2015.

4.4 Technology Summary Table

In order to provide a simple comparison of all operational, mechanical, and ecological technologies researched, a technology summary table was developed and is included in Appendix C. The table is organized by category (operational, mechanical, and ecological) and includes information on major performance, operational, maintenance, and construction criteria that should be used in the selection of an appropriate technology for each site-specific condition. Technologies researched in this table are further evaluated in Section 5.0.

5.0 RECOMMENDED TECHNOLOGIES/BMPS

Section 4.0 includes a detailed literature review of operational, mechanical, and ecological technologies and BMPs available for optimization of facultative lagoons. The goal of this report is to give lagoon operators a "design toolbox" of operational and design alternatives in order to meet the specific needs of their facility. Within Section 4.0, a recommendations section is included for each technology that summarizes the technology and identifies its most appropriate applications. The process for selection of the most appropriate technologies is not a "one size fits all" approach. Each lagoon in Montana has specific site conditions that may suggest the use of certain technologies (available space to land apply effluent) or preclude the use of others (lack of available space prevents addition of wetland cells). Although there are several lagoon technologies that are likely to be successful at most sites (aeration,

fixed film, phosphorus removal, etc.), other factors may limit their implementation (financial capacity of community, power requirements, etc.).

From an operational improvement standpoint, the activities and BMPs detailed in Section 4.1 are generally applicable and should ideally be conducted on all facultative lagoons, as appropriate, in contrast to new technologies that may only be appropriate in some cases. This report simply identifies the various operation and maintenance improvements available. As a result, the implementation of these BMPs should not necessarily be a "selection", but part of an overall evaluation of a facility to determine if they are being conducted. A lagoon operations checklist could be developed to easily identify what operational activities are currently being conducted, and those that could be added to improve performance.

A two-tiered evaluation was developed to create the ability to consider individual criteria and accurately select the appropriate technology/technologies specific to each community's need. Tier 1 consists of more general criteria related to each community (geography, population, effluent conditions), while Tier 2 consists of site specific design conditions that will determine the applicability of some technologies and BMPs. Also included is technology-specific information that may guide lagoon operators in their selection. The tiered evaluation criteria are summarized in Table 17.

Table 17. Facultative Lagoon Technology Evaluation Criteria

Tier 1 Criteria	
Selection Criteria	*Description*
Population/Financial Capacity	Community population and financial capacity will impact the ability to finance lagoon improvements. The ability to leverage grant funding will allow consideration of more treatment options.
Climate	The effectiveness of specific treatments may be dependent upon climate (temperature, precipitation) conditions. High elevation/colder climates may have limited treatment options and can expect lower efficiencies.
Effluent Conditions	Effluent characteristics and discharge volume/timing will have a major impact on technology selection.
Performance Needs	Identification of MPDES permit requirements will affect treatment selection.
Tier 2 Criteria	
Capital Costs	Capital costs of some treatments may limit use by smaller communities.
Operation and Maintenance Costs	Operation and maintenance costs may also limit use by smaller communities.
Expected Removal Efficiency	Expected removal efficiency of selected technology must meet performance needs of community

Appendix A – Numeric Nutrient Criteria for Montana (DEQ-12A) and EPA Approval

DEPARTMENT CIRCULAR

DEQ-12A

Montana Base Numeric Nutrient Standards

GENERAL INTRODUCTION

This circular (DEQ-12A) contains information pertaining to the base numeric nutrients standards (§75-5-103(2), MCA) and their implementation. This information includes the standards' concentration limits, where the standards apply, and their period of application. DEQ-12A is adopted by the Board of Environmental Review under its rulemaking authority in §75-5-301(2), MCA.

Circular DEQ-12B contains information about variances from the base numeric nutrient standards and is a separate document available from the Department. DEQ-12B addresses effluent treatment requirements associated with general nutrient standards variances, as well as effluent treatment requirements for individual nutrient standards variances and to whom these apply. Unlike DEQ-12A, DEQ-12B is not adopted by the Board of Environmental Review; DEQ-12B is adopted by the Department following its formal rulemaking process, pursuant to §75-5-313, MCA.

The Department has reviewed a considerable amount of scientific literature and has carried out scientific research on its own in order to derive the base numeric nutrient standards (see **References** in this circular). Because many of the base numeric nutrient standards are stringent and may be difficult for MPDES permit holders to meet in the short term, Montana's Legislature adopted laws (e.g., §75-5-313, MCA) allowing for the achievement of the standards over time via the variance procedures in Circular DEQ-12B. This approach should allow time for nitrogen and phosphorus removal technologies to improve and become less costly and to allow time for nonpoint sources of nitrogen and phosphorus pollution to be better addressed.

Circular DEQ-12A

JULY 2014 EDITION

1.0 Introduction

Elements comprising Circular DEQ-12A are found below. These elements are adopted by the Montana Board of Environmental Review. The nitrogen and phosphorus concentrations provided here have been set at levels that will protect beneficial uses and prevent exceedences of other surface water quality standards which are commonly linked to nitrogen and phosphorus concentrations (e.g., pH and dissolved oxygen; see Circular DEQ-7 for those standards). The nitrogen and phosphorus concentrations provided here also reflect the intent of the narrative standard at ARM 17.30.637(1)(e) and will preclude the need for case-by-case interpretations of that standard in most cases.

1.1 Definitions

1. **Ecoregion** means mapped regions of relative homogeneity in ecological systems derived from perceived patterns of a combination of causal and integrative factors including land use, land surface form, potential natural vegetation, soils, and geology. See also Endnote 1.

2. **Large river** means a perennial waterbody which has, during summer and fall baseflow (August 1 to October 31 each year), a wadeability index (product of river depth [in feet] and mean velocity [in ft/sec]) of 7.24 ft^2/sec or greater, a depth of 3.15 ft or greater, or a baseflow annual discharge of 1,500 ft^3/sec or greater. See also, Endnote 6.

3. **Total nitrogen** means the sum of all nitrate, nitrite, ammonia, and organic nitrogen, as N, in an unfiltered water sample. Total nitrogen in a sample may also be determined via persulfate digestion or as the sum of total kjeldahl nitrogen plus nitrate plus nitrite.

4. **Total phosphorus** means the sum of orthophosphates, polyphosphates, and organically bound phosphates, as P, in an unfiltered water sample. Total phosphorus may also be determined directly by persulfate digestion.

5. **Wadeable stream** means a perennial or intermittent stream in which most of the wetted channel is safely wadeable by a person during baseflow conditions.

2.0 Base Numeric Nutrient Standards

Table 12A-1 contains the base numeric nutrient standards for Montana's flowing waters. In **Table 12A-1** nutrient standards for wadeable streams are grouped by ecoregion, either at level III (coarse scale) or level IV (fine scale). Following the ecoregional standards is a list of wadeable streams with reach-specific standards. These waterbodies have characteristics disimilar from those of the ecoregions in which they reside and have therefore been provided reach-specific values. **For wadeable streams, the standards should be applied in this order: named stream reach first (if applicable) <u>then</u> level IV ecoregion (if applicable) <u>then</u> level III ecoregion. Table 12A-1** also contains a list of large river segments for which base numeric nutrient standards have been developed. Note that the ecoregional values in **Table 12A-1** do not apply to large rivers within those ecoregions. See Endnote 6 for a list of all large Montana rivers. If a particular large river reach is not listed in **Table 12A-1**, standards for it have not yet been developed.

Table 12A-2 is a placeholder table for future base numeric nutrient standards for Montana's lakes and reservoirs. The Department has not yet developed regional lake criteria, but it is expected that when they are developed they will be grouped by ecoregion. As such, placeholders for future ecoregionally-based criteria are provided in the table. The table also provides for lake-specific standards. The Department anticipates that reservoir standards will generally be developed case-by-case and, therefore, will be individually listed, as provided for in the table.

Table 12A-1. Base Numeric Nutrient Standards for Wadeable Streams in Different Montana Ecoregions. If standards have been developed for level IV ecoregions (subcomponents of the level III ecoregions) they are shown in italics below the applicable level III ecoregion. Individual reaches are in the continuation of this table.

			Numeric Nutrient Standard[4]	
Ecoregion[1,2] (level III or IV) and Number	**Ecoregion Level**	**Period When Criteria Apply[3]**	**Total Phosphorus (µg/L)**	**Total Nitrogen (µg/L)**
Northern Rockies (15)	III	July 1 to September 30	25	275
Canadian Rockies (41)	III	July 1 to September 30	25	325
Idaho Batholith (16)	III	July 1 to September 30	25	275
Middle Rockies (17)	III	July 1 to September 30	30	300
Absaroka-Gallatin Volcanic Mountains (17i)	IV	July 1 to September 30	105	250
Northwestern Glaciated Plains (42)	III	June 16 to September 30	110	1300
Sweetgrass Upland (42l), Milk River Pothole Upland (42n), Rocky Mountain Front Foothill Potholes (42q), and Foothill Grassland (42r)	IV	July 1 to September 30	80	560
Northwestern Great Plains (43) and Wyoming Basin (18)	III	July 1 to September 30	150	1300
River Breaks (43c)	IV	See Endnote 5	See Endnote 5	See Endnote 5
*Non-calcareous Foothill Grassland (43s), Shields-Smith Valleys (43t), Limy Foothill Grassland (43u), Pryor-Bighorn Foothills (43v), and Unglaciated Montana High Plains (43o)**	IV	July 1 to September 30	33	440

*For the Unglaciated High Plains ecoregion (43o), criteria only apply to the polygon located just south of Great Falls, MT.

[1] See Endnote 1

[2] See Endnote 2

[3] See Endnote 3

[4] See Endnote 4

Table 12A-1, Continued. Base Numeric Nutrient Standards for Individual Wadeable Streams (and Wadeable-stream Reaches), and Large-river Reaches.

		Numeric Nutrient Standard[4]	
Individual Stream or Reach Description[2]	**Period When Criteria Apply[3]**	**Total Phosphorus (µg/L)**	**Total Nitrogen (µg/L)**
Wadeable Streams: Clark Fork River basin			
Flint Creek, from Georgetown Lake outlet to the ecoregion 17ak boundary (46.4002, -113.3055)	July 1 to September 30	72	500
Wadeable Streams: Gallatin River basin			
Bozeman Creek, from headwaters to Forest Service Boundary (45.5833, -111.0184)	July 1 to September 30	105	250
Bozeman Creek, from Forest Service Boundary (45.5833, -111.0184) to mouth at East Gallatin River	July 1 to September 30	76	270
Hyalite Creek, from headwaters to Forest Service Boundary (45.5833,-111.0835)	July 1 to September 30	105	250
Hyalalite Creek, from Forest Service Boundary (45.5833,-111.0835) to mouth at East Gallatin River	July 1 to September 30	90	260
East Gallatin River between Bozeman Creek and Bridger Creek confluences	July 1 to September 30	50	290
East Gallatin River between Bridger Creek and Hyalite Creek confluences	July 1 to September 30	40	300
East Gallatin River between Hyalite Creek and Smith Creek confluences	July 1 to September 30	60	290
East Gallatin River from Smith Creek confluence mouth (Gallatin River)	July 1 to September 30	40	300
Large Rivers[6] :			
Yellowstone River (Bighorn River confluence to Powder River confluence)	August 1 -October 31	55	655
Yellowstone River (Powder River confluence to stateline)	August 1 -October 31	95	815

[2] See Endnote 2

[3] See Endnote 3

[4] See Endnote 4

[6] See Endnote 6

Table 12A-2. Base Numeric Nutrient Standards and Other Standards for Lakes and Reservoirs.

		Numeric Nutrient Standard[7]		
Ecoregion[1] (level III) and Number, or Individual Lake or Reservoir Description	**Period of Application**	**Total Phosphorus (µg/L)**	**Total Nitrogen (µg/L)**	**Other Standards[8]**
LAKES/RESERVOIRS by ecoregion:				
Middle Rockies (17)	Year-round	[]	[]	
Northern Rockies (15)	Year-round	[]	[]	
Canadian Rockies (41)	Year-round	[]	[]	
Idaho Batholith (16)	Year-round	[]	[]	
LAKE SPECIFIC CRITERIA:				
	Year-round	[]	[]	
RESERVOIR SPECIFIC CRITERIA:				
	Year-round	[]	[]	

[1] See Endnote 1

[7] See Endnote 7

[8] See Endnote 8

2.1 Required Reporting Values for Base Numeric Nutrient Standards

Table 12A-3 presents the required reporting values (RRVs) for total phosphorus and total nitrogen, as well as the RRVs for nitrogen fractions that can be used to compute total nitrogen.

Table 12A-3. Required reporting values[a,b] for total nitrogen and phosphorus measurements.

Nutrient		Method of Measurement	Required Reporting Value
Total phosphorus		Persulfate digestion	3 µg/L
Total nitrogen		Persulfate digestion	70 µg/L
Total nitrogen	Sum of:	(a) total kjeldahl nitrogen	225 µg/L
		(b) nitrate + nitrite	See RRVs below
Nitrate- as N			20 µg/L
Nitrite- as N			10 µg/L
Nitrate + Nitrite-as N			20 µg/L

[a] See definition for required reporting values found in footnote 19 of Department Circular DEQ-7.

[b] Concentrations in Table 12A-3 must be achieved unless otherwise specified in a permit, approval, or authorization issued by the Department (DEQ-7; ARM 17.30.702).

2.2 Developing Permit Limits for Base Numeric Nutrient Standards

For total nitrogen and total phosphorus, the critical low-flow for the design of disposal systems shall be based on the seasonal 14Q5 of the receiving water (ARM 17.30.635(2)). When developing permit limits for base numeric nutrient standards, the Department will use an average monthly limit (AML) only, based on a calendar month, using methods appropriate for criterion continuous concentrations (i.e., chronic concentrations). Permit limits will be established using a value corresponding to the 95th percentile probability distribution of the effluent. Nitrogen and phosphorus concentrations of the receiving waterbody upstream of the discharge may be characterized using other frequency distribution percentiles. The Department shall use methods that are appropriate for criterion continuous concentrations which are found in the document "*Technical Support Document for Water Quality-based Toxics Control*," Document No. EPA/505/2-90-001, United States Environmental Protection Agency, 1991.

3.0 Endnotes

(1) Ecoregions are based on the 2009 version (version 2) of the U.S. Environmental Protection Agency maps. These can be found at: http://www.epa.gov/wed/pages/ecoregions/mt_eco.htm . For Geographic Information System (GIS) use within the Department, the GIS layers may be found at: L:\DEQ\Layers\Reference\Ecoregions.lyr

(2) Within and among the geographic regions or watersheds listed, base numeric nutrient standards of the downstream reaches or other downstream waterbodies must continue to be maintained. Where possible, modeling methods will be utilized to determine the limitations required which provide for the attainment and maintenance of water quality standards of downstream waterbodies.

(3) For the purposes of ambient surface water monitoring and assessment only, a ten-day window (plus/minus) on the beginning and ending dates of the period when the criteria apply is allowed in order to accommodate year-specific conditions (an early-ending spring runoff, for example).

(4) The average concentration during a period when the standards apply may not exceed the standards more than once in any five-year period, on average.

(5) In this level IV ecoregion, the narrative standard for nuisance aquatic life (ARM 17.30.637(1)(e)) applies in lieu of specific base numeric nutrient standards.

(6) **Table E-1** below shows the beginning and ending locations for large rivers in Montana.

Table E-1. Large river segments within the state of Montana.

River Name	Segment Description
Big Horn River	Yellowtail Dam to mouth
Clark Fork River	Bitterroot River to state-line
Flathead River	Origin to mouth
Kootenai River	Libby Dam to state-line
Madison River	Ennis Lake to mouth
Missouri River	Origin to state-line
South Fork Flathead River	Hungry Horse Dam to mouth
Yellowstone River	State-line to state-line

(7) No lake or reservoir in **Table12A-2** shall have a total nutrient concentration that exceeds the values shown, as an annual average, more than once in any three year period, on average. The Department will determine on a case-by-case basis whether or not a permitted discharge to a stream or river is likely to be affecting any downstream lake or reservoir. If so, the permittee would be required to meet its average monthly nutrient limit year-round.

(8) Parameters listed under this column are standards specific to lakes and reservoirs.

4.0 References

The following are citations for key scientific and technical literature used to derive the base numeric nutrient standards. This is not a complete list; rather, it contains the most pertinent citations. Many other articles and reports were reviewed during the development of the standards.

Biggs, B.J.F., 2000. New Zealand Periphyton Guideline: Detecting, Monitoring and Managing Enrichment in Streams. Prepared for the New Zealand Ministry of the Environment, Christchurch, 122 p.

Dodds, W.K., V.H. Smith, and B. Zander, 1997. Developing Nutrient Targets to Control Benthic Chlorophyll Levels in Streams: A Case Study of the Clark Fork River. Water Research 31: 1738-1750.

Dodds, W.K., V.H. Smith, and K. Lohman, 2002. Nitrogen and Phosphorus Relationships to Benthic Algal Biomass in Temperate Streams. Canadian Journal of Fisheries and Aquatic Sciences 59: 865-874.

Dodds, W.K, V.H. Smith, and K. Lohman, 2006. Erratum: Nitrogen and Phosphorus Relationships to Benthic Algal Biomass in Temperate Streams. Canadian Journal of Fisheries and Aquatic Sciences 63: 1190-1191.

Elser, J.J., M.E.S. Bracken, E.E. Cleland, D.S. Gruner, W.S. Harpole, H. Hillebrand, J.T. Ngai, E.W. Seabloom, J.B. Shurin, and J.E. Smith, 2007. Global Analysis of Nitrogen and Phosphorus

Limitation of Primary Producers in Freshwater, Marine and Terrestrial Ecosystems. Ecology Letters 10: 1135-1142.

Flynn, K., and M.W. Suplee, 2010. Defining Large Rivers in Montana using a Wadeability Index. Helena, MT: Montana Department of Environmental Quality, 14 p.

Flynn, Kyle and Michael W. Suplee. 2013. Using a Computer Water Quality Model to Derive Numeric Nutrient Criteria: Lower Yellowstone River. WQPBDMSTECH-22. Helena, MT: Montana Dept. of Environmental Quality. http://deq.mt.gov/wqinfo/standards/NumericNutrientCriteria.mcpx

McCarthy, P.M., 2005. Statistical Summaries of Streamflow in Montana and Adjacent Areas, Water years 1900 through 2002. U.S. Geological Survey Scientific Investigations Report 2004-5266, 317 p.

Omernik, J.M., 1987. Ecoregions of the Conterminous United States. Annals of the Association of American Geographers 77: 118-125.

Smith, R.A., R.B. Alexander, and G.E. Schwarz, 2003. Natural Background Concentrations of Nutrients in Streams and Rivers of the Conterminous United States. Environmental Science and Technology 37: 3039-3047.

Sosiak, A., 2002. Long-term Response of Periphyton and Macrophytes to Reduced Municipal Nutrient Loading to the Bow River (Alberta, Canada). Canadian Journal of Fisheries and Aquatic Sciences 59: 987-1001.

Stevenson, R.J, S.T. Rier, C.M. Riseng, R.E. Schultz, and M.J. Wiley, 2006. Comparing Effects of Nutrients on Algal Biomass in Streams in Two Regions with Different Disturbance Regimes and with Applications for Developing Nutrient Criteria. Hydrobiologia 561: 149-165.

Suplee, M., R. Sada de Suplee, D. Feldman, and T. Laidlaw, 2005. Identification and Assessment of Montana Reference Streams: A Follow-up and Expansion of the 1992 Benchmark Biology Study. Helena, MT: Montana Department of Environmental Quality, 41 p.

Suplee, M.W., A. Varghese, and J. Cleland, 2007. Developing Nutrient Criteria for Streams: An Evaluation of the Frequency Distribution Method. Journal of the American Water Resources Association 43: 453-472.

Suplee, M.W., V. Watson, A. Varghese, and J. Cleland, 2008. Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers. Helena, MT: Montana Department of Environmental Quality, 86 p. http://deq.mt.gov/wqinfo/standards/NumericNutrientCriteria.mcpx

Suplee, M.W., and V. Watson, 2013. Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers—Update 1, *and addendums*. Helena, MT: Montana Dept. of Environmental Quality. http://deq.mt.gov/wqinfo/standards/NumericNutrientCriteria.mcpx

Suplee, M.W., V. Watson, M. Teply, and H. McKee, 2009. How Green is too Green? Public Opinion of what Constitutes Undesirable Algae Levels in Streams. Journal of the American Water Resources Association 45: 123-140.

Suplee, M.W., and R. Sada de Suplee, 2011. Assessment Methodology for Determining Wadeable Stream Impairment Due to Excess Nitrogen and Phosphorus Levels. Helena, MT: Montana Department of Environmental Quality

Suplee, M.W., V. Watson, W.K. Dodds, and C. Shirley, 2012. Response of Algal Biomass to Large Scale Nutrient Controls on the Clark Fork River, Montana, United States. Journal of the American Water Resources Association 48: 1008-1021.

U.S. Environmental Protection Agency, 2000. Nutrient Criteria Technical Guidance Manual, Rivers and Streams. United States Environmental Protection Agency, EPA-822-B00-002. Washington, D.C.

Varghese, A., and J. Cleland, 2005. Seasonally Stratified Water Quality Analysis for Montana Rivers and Streams-Final Report. Prepared by ICF International for the Montana Department of Environmental Quality, 44 p plus appendices.

Varghese, A., J. Cleland, and B. Dederick, 2008. Updated Statistical Analyses of Water Quality Data, Compliance Tools, and Changepoint Assessment for Montana Rivers and Streams. Prepared by ICF International for the Montana Department of Environmental Quality under agreement No. 205031, task order 5.

Woods, A.J., J.M. Omernik, J.A. Nesser, J. Shelden, J.A. Comstock, and S. J. Azevedo, 2002. Ecoregions of Montana, 2nd edition. (Color Poster with Map, Descriptive Text, Summary Tables, and Photographs): Reston, Virginia, U.S. Geological Survey (map scale 1:1,500,000).

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Tom Livers, Acting Director
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P.O. Box 200901
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Robin Shropshire, Chairman
Montana Board of Environmental Review
Montana Department of Environmental Quality
P.O. Box 200901
Helena, Montana 59620-0901

Re: EPA Action on Montana's Numeric Nutrient Criteria and Variance Rules

Dear Mr. Livers and Ms. Shropshire:

The U.S. Environmental Protection Agency Region 8 has completed its review of Montana's new and revised water quality standards for nutrients and is approving the water quality standards as described in the enclosure. The Montana Department of Environmental Quality (Montana or MDEQ) and the Montana Board of Environmental Review (BER or the Board) adopted these revisions on July 25, 2014, and submitted the revisions to the EPA for review pursuant to 40 CFR Section 131.20(c). The submission included: (1) a copy of the adopted amendments and supporting materials; (2) notice of final adoption of the amendments with the state's response to comments; and (3) a letter certifying that the amendments and water quality standards were adopted in accordance with state law. Receipt of this submission on August 15, 2014, initiated the EPA's review pursuant to Section 303(c) of the Clean Water Act (CWA or the Act) and the federal water quality standards implementing regulation (40 CFR Part 131).

We commend the MDEQ and the BER for adopting protective numeric nutrient criteria for total nitrogen and total phosphorus to address nutrient pollution in Montana's surface waters. Montana's nutrient rules include:

- Adoption of numeric nutrient criteria (referred to as "base numeric nutrient standards" in the state's documents) for wadeable streams (Department Circular DEQ-12A);
- Adoption of numeric nutrient criteria (NNC) for segments of the Yellowstone River (Department Circular DEQ-12A);
- A general variance authorizing provision and general variances for public and private dischargers applicable for up to 20 years to waters with numeric nutrient criteria (Department Circular DEQ-12B); and
- Individual variance procedures applicable to waters with numeric nutrient criteria (Department Circular DEQ-12B).

The adopted water quality criteria and variance provisions that are the subject of today's action are scientifically defensible, well supported by the record and consistent with CWA requirements. The EPA looks forward to continuing to work with Montana to protect and improve surface water quality within the state. As a result of the water quality standards, the EPA expects that concentrations of nutrients in Montana surface waters will decline over time.

Clean Water Act Review Requirements

The CWA Section 303(c)(2) requires states and authorized Indian tribes[1] to submit new or revised water quality standards (WQS) to the EPA for review. The EPA is required to review and approve or disapprove, the submitted standards. The Region's goal has been, and will continue to be, to work closely with states and authorized tribes throughout the standards revision process to help ensure that submitted water quality standards adopted by states are consistent with CWA requirements. Pursuant to 40 CFR Section 131.21(c), new or revised state standards submitted to the EPA after May 30, 2000, are not effective for CWA purposes until approved by the EPA. 65 Fed. Reg. 24653 (April 27, 2000).

Today's Action

Today the EPA is approving a number of water quality standards provisions discussed below, including numeric nutrient criteria and variance provisions. The EPA has concluded that the adopted provisions are consistent with the requirements of the Clean Water Act and the EPA's implementing regulations. The enclosure contains a more detailed rationale for today's action.

Endangered Species Act Requirements

The EPA's approval of Montana's water quality standards is considered a federal action which may be subject to the Section 7(a)(2) consultation requirements of the Endangered Species Act (ESA). Section 7(a)(2) of the ESA states that "each federal agency ... shall ...insure that any action authorized, funded or carried out by such agency is not likely to jeopardize the continued existence of any endangered species or threatened species or result in the destruction or adverse modification of habitat of such species which is determined to be critical..."

The EPA's approval of new or revised water quality standards, therefore, may be subject to the results of consultation with the U.S. Fish and Wildlife Service (Service) pursuant to Section 7(a)(2) of the ESA. Nevertheless, the EPA also has a CWA obligation, as a separate matter, to complete its WQS action. Therefore, in acting on the state's WQS today, the EPA is completing its CWA Section 303(c) responsibilities.

The EPA's approval of the following water quality standards revisions is not subject to ESA consultation because either the actions will have "no effect" on listed aquatic and aquatic-dependent species or the EPA does not have discretion to act upon listed species as discussed in more detail below. All other provisions (i.e., low flow provisions, numeric nutrient criteria, the general variances, individual variance provisions) are approved by the EPA today subject to ESA consultation.

[1] CWA Section 518(e) specifically authorizes EPA to treat eligible Indian tribes in the same manner as states for purposes of CWA Section 303. See also 40 CFR Section 131.8.

No effect revisions

- New Definitions
  - The new definitions are consistent with the EPA's regulations and guidance and support the new Department Circular DEQ-12A. The EPA has determined that its approval of the new definitions will not change the existing environmental conditions. Therefore, ESA consultation is not required.
- Non-substantive edits
  - The EPA considers non-substantive edits to existing WQS to constitute new or revised WQS to ensure public transparency.[2] Montana adopted several revisions that would be included in this category. These revisions do not substantively change the meaning or intent of the existing WQS; therefore, the EPA has determined that these revisions will have no effect on listed species.
- Individual and general variance authorizing provisions
  - ARM 17.30.660(1) is merely an authorizing policy (40 CFR § 131.13) and thus has no effect on listed or proposed endangered or threatened species or critical habitat. As a result, no consultation is required.

No discretion revisions

- Antidegradation revisions
  - Montana revised their existing antidegradation rule ("nondegradation rule") to consider nutrients as a "harmful" parameter for nondegradation purposes instead of as "toxic". The basis for the EPA's conclusion that approval of antidegradation revisions is not subject to ESA consultation is discussed in "Antidegradation Policy Approvals and Endangered Species Act Consultations." Memorandum from Geoff Grubbs, Director, Office of Science and Technology, to Water Management Division Directors, Regions 1 - 10, January 27, 2005. Since the MT antidegradation revisions meet the EPA's regulatory requirements, the EPA has no relevant discretion for ESA purposes.

Indian Country

The WQS approvals in today's letter apply only to waterbodies in the state of Montana, and do not apply to waters that are within Indian country, as defined in 18 U.S.C. Section 1151. "Indian country" includes any land held in trust by the United States for an Indian tribe and any other areas defined as "Indian country" within the meaning of 18 U.S.C. Section 1151. Today's letter is not intended as an action to approve or disapprove water quality standards applying to waters within Indian country. The EPA, or authorized Indian tribes, as appropriate, will retain responsibilities for water quality standards for waters within Indian country.

Conclusion

The EPA Region 8 thanks MDEQ and the Board for their efforts to develop and adopt numeric nutrient criteria for Montana. The nutrient criteria and variance provisions represent significant progress towards addressing nutrient pollution issues in the state. The EPA looks forward to working with MDEQ to

[2] See EPA's October 2012 *What is a New or Revised Water Quality Standard Under CWA 303(c)(3)?-- Frequently Asked Questions* available at http://water.epa.gov/scitech/swguidance/standards/cwa303faq.cfm.

make additional improvements to the state's water quality standards in the future. If you have any questions, please call Tina Laidlaw on my staff at (406) 457-5016.

Sincerely,

Martin Hestmark
Assistant Regional Administrator
Office of Ecosystems Protection
and Remediation

Enclosures

cc: George Mathieus, Division Administrator
Montana Department of Environmental Quality

Connie Howe
Crow Tribe
(via email)

Charlene Alden
Northern Cheyenne Tribe
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Confederated Salish and Kootenai Tribe
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Rationale for the EPA's Action on Montana's New and Revised Water Quality Standards

Today's EPA action letter addresses Montana's new and revised water quality standards for nutrient pollution adopted by the Board and MDEQ on July 25, 2014, including revisions made to Administrative Rules of Montana (ARM) Title 17, Chapter 30 (Water Quality), Sub-chapters 5 (Mixing Zones), 6 (Surface Water Quality Standards and Procedures), and 7 (Nondegradation) as well as adoption of new Department Circulars DEQ-12A and -12B.[3] This enclosure provides a rationale for the action taken by the EPA.

NONSUBSTANTIVE CHANGES TO EXISTING WATER QUALITY STANDARDS

The EPA considers non-substantive edits to existing water quality standards to constitute new or revised water quality standards that the EPA has the authority and duty to approve or disapprove under CWA Section 303(c)(3).[4] Montana adopted several revisions that would be included in this category such as: spelling corrections; adding or removing the word "and"; or numbering changes. The list below identifies those revisions that the EPA considers as non-substantive changes to water quality standards. While these revisions do not substantively change the meaning or intent of the existing water quality standards, the EPA believes it is reasonable to treat such non-substantive changes in this manner to ensure public transparency of which provisions are effective for CWA purposes. Accordingly, all non-substantive revisions to the ARM (Sections 17.30.201(6)(f); 17.30.507(1); 17.30.516(3); 17.30.619(1)(c) and (d); 17.30.619(3); 17.30.622(3)(h) and (i); 17.30.623(2)(h) and (i); 17.30.624(2)(h) and (i); 17.30.625(2)(h) and (i); 17.30.626(2)(h) and (i); 17.30.627(2)(h) and (i); 17.30.628(2)(j) and (k); 17.30.629(2)(h) and (i); 17.30.702; 17.30.702(17) through (20); 17.30.702(22); 17.30.702(27)(c) through (e); 17.30.702(25) and (26); and 17.30.702(27)(c) through (e); 17.30.715(h) are approved.

DEFINITIONS

Montana's nutrient pollution rules include the following definitions:

- Section 1.1 of Department Circular DEQ-12A includes definitions for the following terms: ecoregion, large river, total nitrogen, total phosphorus, and wadeable stream.
- ARM Sections 17.30.602(33) and 17.30.702(23) include revisions to the methods for calculating total nitrogen (TN) concentrations. The language cites the persulfate digestion method for determining total nitrogen and specifies the nutrient fractions (i.e., nitrate, nitrite, ammonia, and organic nitrogen, as N) that can be summed to calculate the total nitrogen concentration. ARM Sections 17.30.602(34) and 17.30.702(24) include similar revisions to the definitions for total phosphorus.
- ARM Sections 17.30.602(39), 17.30.619(1)(a), and 17.30.702(27)(a) modify the reference to nutrient standards previously contained in Circular DEQ-7. Water quality standards for nutrients (total nitrogen (TN) and total phosphorus (TP)) are now contained in Circular DEQ-12A. Human health-based water quality standards for nitrate, nitrate + nitrite, and nitrite, which have toxic effects, will remain in Circular DEQ-7.

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[3] Department Circular DEQ-12A and Department Circular DEQ-12B have been incorporated by reference into Montana's existing water quality standards at ARM 17.30.507(1)(a); 17.30.619(1)(e); 17.30.660(1); and 17.30.660(8) which provides additional assurances that these Circulars are legally binding.

[4] See EPA's October 2012 *What is a New or Revised Water Quality Standard Under CWA 303(c)(3)?*- Frequently Asked Questions available at http://water.epa.gov/scitech/swguidance/standards/cwa303faq.cfm.

- ARM Sections 17.30.602(40) and 17.30.702(27)(b) include a description of Circular DEQ-12A ("Montana Base Numeric Nutrient Standards"). Circular DEQ-12A contains Montana's adopted numeric nutrient criteria (NNC) for TN and TP.
- ARM Section 17.30.602(41) includes a reference to Department Circular DEQ-12B ("Montana Base Numeric Nutrient Standards Variances"). Circular DEQ-12B describes the requirements for the general variances for nutrients and the procedures for obtaining an individual nutrient variance. Any future approved individual variances will be contained in Circular DEQ-12B.
- ARM Section 17.30.702(17) was repealed because the definition of "nutrients" as inorganic nitrogen and inorganic phosphorus does not align with the numeric criteria adopted in Department Circular DEQ-12A for total nitrogen and total phosphorus.

The EPA has reviewed these definitions and considers them to be scientifically sound and consistent with the requirements of 40 CFR Part 131 as discussed below. Therefore, these provisions are approved.

CRITICAL LOW FLOW PROVISIONS

Section 2.2 in Department Circular DEQ-12A and revisions to Sections ARM 17.30.516 (3)(e) and (4) and ARM 17.30.635(2) identify critical low flows for purposes of calculating water quality-based effluent limitations (WQBELs) for nutrients to be included in CWA National Pollutant Discharge Elimination System (NPDES) permits.

ARM 17.30.516(3):

(e) Facilities that discharge the parameters found in Department Circular DEQ-12A to surface water. Discharge limitations must be based on dilution with the entire seasonal 14-day, five-year (seasonal 14Q5) low flow of the receiving water without the discharge.

ARM 17.30.635: General Treatment Standards

(2) For total nitrogen and total phosphorus, the stream flow dilution requirements must be based on the seasonal 14Q5, which is the lowest average 14 consecutive day low flow, occurring from July through October, with an average recurrent frequency of once in five years.

ARM 17.30.516(4) specifies that, for nutrients only, mixing zone determinations are based on the seasonal 14Q5 low flow.

Montana typically uses a 7Q10 (seven-day, ten-year design flow) as the critical low flow for determining the allowable permitted discharge for toxics and other parameters. Since nutrients (i.e., TN, TP) are generally not toxic, Montana explored different options for selecting the critical low flow and determined that a seasonal 14-day, 5-year design flow was appropriate for discharges containing nutrients. The basis for the low flow provisions is described in a memo to the BER.[5] Montana used algal growth rates derived from laboratory studies to model the time (measured in days) it would take to reach peak algal biomass in a stream. Applying the model, the state estimated the number of days it would take before algal biomass concentrations reached nuisance bloom levels of 150 mg/m^2.[6] Results showed that peak algal biomass was achieved in 14-days, on average. However, depending on the initial biomass used in the model, this estimate could be over or under protective. Therefore, Montana compared the

[5] Memo from Mike Suplee and Kyle Flynn, MDEQ, to the Board of Environmental Review, 19 March 2014.

[6] Suplee, M.W;V. Watson, M.E. Teply, and H. McKee. 2009. How Green is Too Green? Public Opinion of what Constitutes Undesirable Algae Levels in Streams. Journal of the American Water Resources Association 45: 123-140.

proposed duration to results from the whole-stream nutrient enrichment study conducted in eastern Montana. Results from that study showed that peak biomass was reached approximately 20 days after the start of the nutrient additions. This comparison validated Montana's selection of a 14-day duration low flow period associated with the NNC.

Basis for Approval

The EPA's water quality standards regulation explains that "States may, at their discretion, include in their State standards, policies generally affecting their application and implementation, such as mixing zones, low flows and variances. Such policies are subject to EPA review and approval (40 CFR § 131.13)." The revision to Montana's low flow provisions for nutrients identifies river and stream low flows, for use in calculating nutrient WQBELs, which are consistent with the adopted NNC. Montana's NNC are average growing season concentrations that cannot be exceeded more than once in every five years. The EPA reviews low flow provisions to ensure they are consistent with the duration and frequency provisions of the criterion. Montana selected a 14Q5 low flow provision that is shorter in duration (and therefore protective) than the NNC which are expressed as seasonal average criteria. Therefore, the EPA finds that Montana's low flow provision is appropriate and will support WQBELs that derive from and comply with the NNC.

The EPA concludes that Montana's low-flow provisions are appropriate because the duration and frequency of the flows support calculation of WQBELs that derive from and comply with the NNC.[7] (See 40 CFR § 131.11, 40 CFR § 131.13). Accordingly, the EPA approves these provisions.

ANTIDEGRADATION

Montana removed the term "nutrients" from ARM 17.30.715(c) and revised ARM 17.30.715(f) to include the parameters listed in DEQ-12-A (TN and TP). The practical effect of this revision is that it changes the nonsignificance threshold that applies to TN and TP from the 15% of the lowest applicable standard that applies to "toxic" parameters, to the one that applies to "harmful" parameters which is 10% of the applicable standard and existing water quality less than 40% of the standard. The state did not change the nonsignificance thresholds that apply to toxic or harmful parameters, it simply reclassified TN and TP from toxic to harmful.

Basis for Approval

The EPA's WQS regulation requires states to adopt an antidegradation policy and identify implementation procedures that at a minimum are consistent with 40 CFR § 131.12(a)(1-4). As described in the EPA Water Quality Standards Handbook (1994), "EPA's review of the implementation procedures is limited to ensuring that procedures are included that describe how the State will implement the required elements of the antidegradation review. The EPA may disapprove and federally promulgate all or part of an implementation process for antidegradation if, in the judgment of the Administrator, the State's process (or certain provisions thereof) can be implemented in such a way as to circumvent the intent and purpose of the antidegradation policy."

The EPA has reviewed the revisions to ARM 17.30.715(1)(c) and (f) and determined that they do not undermine the intent and purpose of Montana's nondegradation policy. Changing the significance test

[7] The EPA guidance on critical low flow provisions is available on the website at: http://water.epa.gov/scitech/swguidance/standards/handbook/chapter05.cfm#section52.

that applies to TN and TP from toxic to harmful continues to protect assimilative capacity for these parameters where it exists, which is clearly consistent with the intent and purpose of the nondegradation policy.

In addition, the environmental effects of TN and TP are not consistent with Montana's definition of the term "toxic". Montana defines a "toxic" parameter as: "A toxin is any chemical which has an immediate, deleterious effect on the metabolism of a living organism."[8] In contrast, the environmental effects of elevated levels of nitrogen and phosphorus may include excess algal growth; lower dissolved oxygen concentrations or increased fluctuations in dissolved oxygen and pH; decreased water clarity; and loss of sensitive species.

The EPA concludes these revisions are consistent with 40 CFR § 131.12 and are approved.

NONSEVERABILITY PROVISION

Montana included in its regulations (ARM 17.30.619(2) and 17.30.715(4)) a provision that calls for the voiding of *all* adopted NNC and all variances should one of three triggering events occur. The EPA is committed to continuing its collaboration with the state to implement this nutrient rule approach consistent with CWA requirements, including the adoption of variances established by and consistent with ARM 17.30.660 and Montana Circular DEQ-12B. Thus, the EPA believes it was inadvisable for the state to include such a provision. The EPA is not acting on this provision today.

NUMERIC NUTRIENT CRITERIA

Clean Water Act requirements relating to Numeric Nutrient Criteria

In reviewing water quality criteria, the EPA determines whether the criteria protect the designated use and are based on a sound scientific rationale. *See* 40 CFR § 131.5(a)(2), (5); 131.6(b)-(c) and 131.11(a). The regulations also require that for waters with multiple use designations, the criteria shall support the most sensitive use. 40 CFR § 131.11(a). As discussed below, the EPA has determined that Montana's NNC adopted in DEQ-12A are consistent with CWA requirements.

EPA Recommendations on Deriving Numeric Nutrient Criteria

For over a decade, the EPA has recognized the importance of developing numeric water quality criteria to protect the designated uses of waterbodies from nutrient pollution that is associated with increases in concentrations of nitrogen and phosphorus. In general, the EPA recommends three types of scientifically defensible approaches for setting numeric criteria to address nitrogen and phosphorus pollution: reference condition approach, stressor-response analysis, and mechanistic modeling.[9,10] The reference condition approach relies on data collected at minimally disturbed reference sites to characterize natural background conditions using percentiles of the frequency distribution from the reference dataset.

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[8] Montana DEQ, Planning Prevention and Assistance Division, Water Quality Planning Bureau, Water Quality Standards Section. 2012. DEQ-7 Montana Numeric Water Quality Standards. Helena, MT: Montana Dept. of Environmental Quality.

[9] U.S. EPA. 2000. Nutrient Criteria Technical Guidance Manual: Rivers and Streams. EPA-822-B-00002. http://water.epa.gov/scitech/swguidance/standards/criteria/nutrients/rivers/index.cfm. Washington, DC.

[10] U.S. EPA. 2010. Using Stressor-response Relationships to Derive Numeric Nutrient Criteria. EPA-820-S-10-001. Washington, DC.

Deriving nutrient criteria using stressor-response analysis provides an empirical representation of the known causal relationship between increased nutrients and ecological effects. In this approach, the known causal relationship has been established in the scientific literature by observational and manipulative studies. Mechanistic modeling refers to use of watershed models, hydrodynamic models or water quality models to determine NNC. A modeling approach to setting nutrient criteria allows the user to test the interactions between different nutrient loading scenarios, the response endpoint(s), and the candidate nutrient criteria. As discussed in detail below, Montana used a combination of reference and stressor-response approaches that is consistent with the EPA's recommendation to derive the NNC for nitrogen and phosphorus and therefore EPA has concluded that Montana's NNC are based on sound science.

Water Quality Standards: Department Circular DEQ-12A Sections 2.0 and 3.0:

Montana promulgated nutrient water quality standards including numeric criteria for total nitrogen and total phosphorus for all wadeable streams, segments of the Yellowstone River, and site-specific nitrogen and phosphorus criteria for several segments in the Gallatin watershed. Table 12A-1 of Circular DEQ-12A Section 2.0 (Table 1) summarizes the NNC approved by the BER and defines the index period when the criteria apply.

Table 1. Montana's Numeric Criteria for TN and TP for Wadeable Streams

Table 12A-1. Base Numeric Nutrient Standards for Wadeable Streams in Different Montana Ecoregions. If standards have been developed for level IV ecoregions (subcomponents of the level III ecoregions) they are shown in italics below the applicable level III ecoregion. Individual reaches are in the continuation of this table.

			Numeric Nutrient Standard[4]	
Ecoregion[1,2] (level III or IV) and Number	**Ecoregion Level**	**Period When Criteria Apply[3]**	**Total Phosphorus (µg/L)**	**Total Nitrogen (µg/L)**
Northern Rockies (15)	III	July 1 to September 30	25	275
Canadian Rockies (41)	III	July 1 to September 30	25	325
Idaho Batholith (16)	III	July 1 to September 30	25	275
Middle Rockies (17)	III	July 1 to September 30	30	300
Absaroka-Gallatin Volcanic Mountains (17i)	IV	July 1 to September 30	105	250
Northwestern Glaciated Plains (42)	III	June 16 to September 30	110	1300
Sweetgrass Upland (42l), Milk River Pothole Upland (42n), Rocky Mountain Front Foothill Potholes (42q), and Foothill Grassland (42r)	IV	July 1 to September 30	80	560
Northwestern Great Plains (43) and Wyoming Basin (18)	III	July 1 to September 30	150	1300
River Breaks (43c)	IV	See Endnote 5	See Endnote 5	See Endnote 5
*Non-calcareous Foothill Grassland (43s), Shields-Smith Valleys (43t), Limy Foothill Grassland (43u), Pryar-Bighorn Foothills (43v), and Unglaciated Montana High Plains (43o)**	IV	July 1 to September 30	33	440

*For the Unglaciated High Plains ecoregion (43o), criteria only apply to the polygon located just south of Great Falls, MT.

Derivation of the Wadeable Streams Nutrient Criteria Based on Omernik[11] Ecoregions

Montana evaluated several approaches (e.g., lithologic groupings, stream order) to characterize the natural variability in nutrient concentrations before selecting Omernik level III ecoregions as the preferred classification scheme. The state's analysis showed statistically significant differences in median nutrient concentrations between level III and level IV ecoregions. However, data limitations precluded establishment of NNC at a finer scale (Omernik level IV) on a statewide basis. The state's analysis and the EPA guidance[12] support Montana's decision to derive NNC at the ecoregion level III scale as being scientifically sound.

Montana followed a multi-step process to establish numeric criteria for TN and TP for wadeable streams. Aquatic life use support was identified as the most sensitive use. By establishing NNC that protect the most sensitive use, Montana's NNC also ensure protection of other designated uses such as recreational use support and drinking water.

1. Montana first characterized nutrient concentrations at reference sites where the aquatic life use was met located within the level III ecoregion.

2. Next, Montana reviewed dose-response studies that were conducted within similar ecoregions and documented in the scientific literature. For each study, Montana identified the nutrient threshold associated with the response endpoint (e.g., algal biomass, diatom or macroinvertebrate metric).

3. Montana used the information obtained from these two approaches (reference and dose-response) as multiple lines of evidence to establish numeric criteria for nitrogen and phosphorus for that ecoregion. Preliminary nutrient criteria were selected using a combination of nutrient percentiles observed at reference sites coupled with thresholds obtained from the relevant stressor-response studies.

4. As a final step in the process, Montana evaluated the nitrogen to phosphorus ratio (N:P ratio / Redfield ratio) associated with the adopted criteria to ensure it was similar to N:P ratios observed at reference sites. N:P ratios can indicate whether nitrogen, phosphorus, or both, are the are the "limiting nutrient" (nutrient in short supply) that constrains algal growth. This final "check" on the proposed criteria ensures that the NNC do not inadvertently alter the limiting nutrient, causing a naturally N-limited stream to become P-limited (or vice versa).

For sites where data were readily available to support the use of level IV ecoregions, Montana established numeric criteria for TN and TP. Examples of level IV ecoregional criteria for TN and TP include (1) the Absaroka-Gallatin Volcanic Mountains where natural background nutrient concentrations are higher than the ecoregion level III nutrient criteria and (2) several level IV ecoregions that reflect transition zones from the mountains to the plains (e.g., Sweetgrass Upland, Pryor-Bighorn Foothills). If dose-response studies were not available for these smaller areas, Montana examined the nutrient concentrations observed in the reference distribution and used the nutrient to benthic chlorophyll-a relationship to calculate the final criteria.

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[11] Omernik, J.M. Ecoregions of the Conterminous United States. *Ann Assoc Am Geogr* 77, 118-125 (1987).
[12] U.S. EPA. 2000. Nutrient Criteria Technical Guidance Manual: Rivers and Streams. EPA-822-B-00002. http://water.epa.gov/scitech/swguidance/standards/criteria/nutrients/rivers/index.cfm. Washington, DC.

Scientific justification for Montana's approach can be found in the May 2013 *Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers,*[13] along with an earlier version of the document published in 2008.[14] Section 3 of Montana's 2013 technical rationale synthesizes the information used to derive the numeric criteria in a concise and easy-to-follow format. For each ecoregion, the document presents: (1) an ecoregional map; (2) recommended numeric criteria; (3) regional reference population descriptive statistics; (4) comparison of the recommended criteria to the ecoregional reference distribution; (5) summary of any relevant dose-response studies; and (6) a conclusion section containing a brief rationale justifying the recommended ecoregional criteria and an evaluation of N:P ratios.

In its scientific justification, Montana recognizes that the ecoregionally-derived nutrient criteria may need to be refined to reflect site-specific considerations, especially in situations where it can be demonstrated that natural background nutrient concentrations exceed the state's ecoregional nutrient criteria and designated uses are supported. To facilitate development of site-specific criteria, Montana described several approaches for deriving site-specific criteria in Section 6.0 of their implementation guidance.[15] Methods include empirically-derived site-specific criteria based on a robust suite of causal and response variable data, or use of a mechanistic model to set protective criteria. The EPA looks forward to working with the state when the state develops such new or revised criteria in the future.

For all NNC adopted by Montana for wadeable streams and rivers, Department Circular DEQ-12A defines the duration and frequency associated with the standard as: "The average concentration during a period when the standards apply may not exceed the standards more than once in any five-year period, on average." (Section 3.0, Endnote 4)

Basis for Approval

Based on review of the Montana's 2008 and 2013 scientific rationales and the comments and technical information submitted to the BER during the state's rulemaking process, the EPA has concluded that the NNC are consistent with CWA requirements discussed above.

In deriving NNC for wadeable streams, Montana independently applied two of the EPA-recommended approaches for deriving NNC (i.e., reference, stressor-response) to build a sound scientific justification for the adopted criteria. In reviewing Montana's scientific rationale, the EPA examined the multiple lines of evidence considered by Montana in establishing the NNC for wadeable streams. Nutrient information gathered from a comprehensive statewide network of reference sites provided useful information on natural background nutrient concentrations observed across Montana. Additionally, the EPA worked closely with Montana to assist the state with developing a rigorous approach to identifying a network of reference sites that represent minimally disturbed reference conditions of aquatic life designated uses. Montana documented their reference screening approach and reference site selection criteria in the 2005 document, "*Identification and Assessment of Montana Reference Streams: A Follow-up and Expansion of the 1992 Benchmark Biology Study*".

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[13] Suplee, M. W., and V. Watson, 2013. Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers—Update 1. Helena, MT: Montana Dept. of Environmental Quality.

[14] Suplee, Michael W., V. Watson, A. Varghese, and Joshua Cleland. 2008. Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers. Helena, MT: MDEQ Water Quality Planning Bureau.

[15] Montana Department of Environmental Quality, 2014. Base Numeric Nutrient Standards Implementation Guidance. Version 1.0. Helena, MT. Montana Dept. of Environmental Quality.

Incorporation of nutrient thresholds identified in regionally relevant dose-response studies further strengthened the state's technical basis for establishing criteria. The state's presentation of the scientific literature provided a sound scientific justification of thresholds associated with impacts to aquatic life and recreational uses observed in studies conducted by academicians, state agencies and other governmental entities (e.g., U.S. Geological Survey). In the 2008 technical basis for the NNC, several peer reviewers (including the EPA) noted the lack of nutrient enrichment studies associated with plains streams. To address these concerns, Montana designed and implemented a whole-stream nutrient addition study on a reference stream in eastern Montana.[16] The purpose of the study was to evaluate the impacts to aquatic life associated with excess algal growth from elevated nutrient levels. Montana used the results from this study to identify stressor-response thresholds for plains streams. The study provided a tremendous amount of useful information that Montana considered in deriving the adopted NNC for plains streams. In addition, the information gathered from Montana's dose-response study strengthened the scientific basis for establishing NNC in plains streams based on stressor-response analysis.

Throughout Montana's NNC development process, the EPA reviewed the state's draft technical documents and provided written comments as well as informal feedback. The EPA also conducted an external independent peer review of the state's preliminary technical rationales for wadeable streams produced in 2008 and 2012. Overall, the peer reviews demonstrated support for Montana's approach as a scientifically sound and defensible basis for developing NNC in wadeable streams. Peer review comments and Montana's response to the comments can be found in the state's technical rationale.[17,18]

The EPA examined Montana's synthesis of the technical basis for the adopted NNC for each ecoregion. For each ecoregion, Montana presented the reference information in addition to the relevant stressor-response studies and offered a detailed and transparent discussion of the basis for the adopted criteria. Montana's integration of multiple approaches -- results from stressor-response studies; understanding of reference conditions; nutrient limitations -- minimizes the uncertainty associated with a single approach and further strengthens the technical basis for the final NNC values.

Therefore, the EPA has determined that the NNC provisions are consistent with the federal requirements because, as discussed above, the state has demonstrated that the NNC for wadeable streams will protect aquatic life and recreational designated uses and are based on a sound scientific rationale that is consistent with the EPA guidance on deriving NNC using scientifically defensible methods. Accordingly, the EPA approves Montana's NNC.

Derivation of Nutrient Criteria for the Yellowstone River

In order to derive NNC for the lower Yellowstone River, Montana chose to utilize an enhanced mechanistic model (QUAL2K). Given the complexity and unique characteristics of large river systems like the Yellowstone, as well as the challenges with determining reference condition for large rivers, Montana determined that utilization of the QUAL2K model to simulate benthic algal growth in the river would be a scientifically defensible approach.

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[16] Suplee, M.W., and R. Sada de Suplee. 2011 Assessment Methodology for Determining Wadeable Stream Impairment Due to Excess Nitrogen and Phosphorus Levels. Helena, MT: Montana Dept. of Environmental Quality. See Appendix B.1.2.
[17] Suplee, M. W., and V. Watson, 2013. Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers—Update 1. Helena, MT: Montana Dept. of Environmental Quality.
[18] See Peer Review Memorandum of 2008 document available at: http://www.deq.mt.gov/wqinfo/standards/NumericNutrientCriteria.mcpx.

Mechanistic modeling is an additional approach recommended by the EPA for establishing defensible NNC. Mechanistic models integrate nutrient-sensitive assessment endpoints and water quality targets to derive protective NNC. Montana spent considerable time and resources to collect the necessary suite of data needed to calibrate and validate the model. Model development is described in more detail below.

After calibrating the model, Montana ran a series of modeling scenarios to simulate the effect of increasing nutrient concentrations on different eutrophication response endpoints associated with impacts to aquatic life, drinking water, and recreational use support (e.g., pH, dissolved oxygen (DO), benthic chlorophyll, total organic carbon, total dissolved oxygen gas). Model simulations of nutrient additions showed that the most sensitive response endpoints (associated with different designated uses) varied between the upper and lower river reaches. Montana then derived the TN and TP criteria necessary to protect the most sensitive use for each segment. For the upper segment of the Yellowstone River (Big Horn River confluence to Powder River confluence), pH was the most sensitive endpoint, indicating that aquatic life use is the most sensitive use. In contrast, for the lower river (Powder River confluence to the state line), the benthic chlorophyll-a threshold (150 mg/m^2) associated with recreational use impacts was the most sensitive response endpoint. As a final step, Montana compared the final numeric criteria to nutrient concentrations in the scientific literature where observed impacts to similar response endpoints have been documented.[19]

Table 2. Numeric Nutrient Criteria for the Yellowstone River

		Numeric Nutrient Standard*	
Individual Stream or Reach Description	Period When Criteria Apply	Total Phosphorus (µg/L)	Total Nitrogen (µg/L)
Yellowstone River (Bighorn River confluence to Powder River confluence)	August 1-October 31	55	655
Yellowstone River(Powder River confluence to stateline)	August 1-October 31	95	815

*The average concentration during a period when the standards apply may not exceed the standards more than once in any five-year period, on average.

Basis for Approval

In reviewing the TN and TP criteria for the segments of the Yellowstone River, the EPA examined the modeling details including: calibration and validation results; simulated response endpoints used to set the criteria; modeled nutrient addition scenarios; design flow; and model uncertainty. Montana tested different simulated response endpoints to confirm that the adopted criteria were protective of the most sensitive use, which for the Yellowstone River included both aquatic life use (upper segment) and recreational use support (lower segment). The EPA reviewed the response indicators applied in the model; model assumptions; and uncertainty factors considered in establishing thresholds. From the review, EPA confirmed the model was developed from a robust dataset; is well calibrated; and accurately simulates nutrient effects on response endpoints. The EPA therefore concludes that the

[19] Montana's detailed scientific basis for TN and TP criteria for segments of the mainstem Yellowstone River can be found in the May 2013 document "*Using a computer water quality model to derive numeric nutrient criteria: Lower Yellowstone River.*"

application of Montana's model for the Yellowstone River produced NNC that are scientifically defensible and protective of designated uses.

In addition to the EPA's internal review, the Agency conducted an external independent peer review of the state's preliminary modeling report describing the scientific basis for the adopted numeric criteria for the mainstem Yellowstone River.[20] Montana responded to reviewer comments in the final report and addressed many of technical issues noted in the comments.

Based on the EPA's review of the technical rationale developed by Montana, the EPA has concluded that the adopted NNC provisions are consistent with 40 CFR § 131.11(a)(1) of EPA's water quality standards regulation. The EPA approves Montana's NNC for the Yellowstone River.

Reach-Specific Criteria: Gallatin Watershed

In addition to the ecoregionally-derived nitrogen and phosphorus criteria for wadeable streams, Department Circular DEQ-12A includes site-specific nutrient criteria for one waterbody in the Clark Fork River basin and eight stream segments in the Gallatin watershed. See Table 1.
For the eight stream segments in the Gallatin watershed, Montana refined the numeric criteria for TN and TP to reflect the contributions of known geologic sources of phosphorus associated with Phosphoria deposits.[21] Portions of the two main tributaries to the Gallatin River, Bozeman and Hyalite Creek, are located within the level IV Absaroka-Gallatin-Volcanic Mountains ecoregion. Montana established level IV nutrient criteria for this area to reflect the naturally elevated total phosphorus concentrations found in these watersheds.[22]

Reach-specific criteria for the tributaries to the Gallatin watershed were calculated using a simple mixing equation to apply in specific locations situations (see below). Natural background (NB) represents the 75th percentile nutrient concentration observed in the reference population from the different contributing ecoregions.[23] This concentration (NB) is multiplied by the average summer flows (Q) for each ecoregional zone to reflect the relative contribution from each area.

$$NB_{NEW} = \frac{(NB_1 * Q_1) + (NB_2 * Q_2)}{Q_1 + Q_2}$$

Following this process, Montana derived reach-specific criteria for Bozeman and Hyalite Creek (See Table 1).[24]

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[20] Peer review comments and Montana's response to the comments can be found in the state's technical rationale: Flynn, Kyle and Michael W. Suplee. 2013. Using a computer water quality model to derive numeric nutrient criteria: Lower Yellowstone River. WQPBDMSTECH-22. Helena, MT: Montana Dept. of Environmental Quality.
[21] Scientific justification for MDEQ's approach can be found on pages 4-4 to 4-8 of the May 2013 document: Suplee, Michael W., V. Watson, A. Varghese, and Joshua Cleland. 2008. *Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers*. Helena, MT: MDEQ Water Quality Planning Bureau.
[22] Id.
[23] The 75th percentile is consistent with EPA's guidance on establishing nutrient criteria for rivers and streams. http://water.epa.gov/scitech/swguidance/standards/criteria/nutrients/rivers/index.cfm.
[24] Id.

Table 1. Reach Specific Nutrient Criteria for the Gallatin River Basin

		Numeric Nutrient Standard*	
Individual Stream or Reach Description	**Period When Criteria Apply**	**Total Phosphorus (µg/L)**	**Total Nitrogen (µg/L)**
Wadeable Streams: Gallatin River Basin			
Bozeman Creek, from headwaters to Forest Service Boundary (45.5833, -111.0184)	July 1 to September 30	105	250
Bozeman Creek, from Forest Service Boundary (45.5833, -111.0184) to mouth at East Gallatin River	July 1 to September 30	76	270
Hyalite Creek, from headwaters to Forest Service Boundary (45.5833, -111.0835)	July 1 to September 30	105	250
Hyalite Creek, from Forest Service Boundary (45.5833, -111.0835) to mouth at East Gallatin River	July 1 to September 30	90	260
East Gallatin River, between Bozeman Creek and Bridger Creek confluences	July 1 to September 30	50	290
East Gallatin River, between Bridger Creek and Hyalite Creek confluences	July 1 to September 30	40	300
East Gallatin River, between Hyalite Creek and Smith Creek confluences	July 1 to September 30	60	290
East Gallatin River, between Smith Creek confluence to mouth (Gallatin River)	July 1 to September 30	40	300

*The average concentration during a period when the standards apply may not exceed the standards more than once in any five-year period, on average.

Basis for Approval

The EPA's water quality standard regulation gives states the discretion and flexibility to establish site-specific criteria that reflect site-specific conditions (40 CFR § 131.11(b)(1)) so long as the criteria protect the designated use and are based on a sound scientific justification. In addition, the Agency produced a memo indicating that states may establish site-specific numeric aquatic life criteria by setting the criteria value equal to natural background.[25]

The EPA has reviewed Montana's reach-specific criteria derived for stream segments in the Gallatin watershed and determined that the criteria reflect natural background conditions associated with phosphorus-rich geologic formations based on nutrient concentrations observed at reference sites from the contributing ecoregions.[26] The Agency also conducted an external independent peer review of the state's preliminary technical rationales for wadeable streams produced in 2012, specifically asking reviewers to comment on the state's proposed approach to deriving reach-specific criteria. Peer review comments considered Montana's approach sound and defensible.

[25] See Memorandum from Tudor T. Davies, Director Office of Science and Technology, Subject: Establishing Site-Specific Aquatic Life Criteria Equal to Natural Background, November 5, 1997.

[26] Scientific justification for MDEQ's approach can be found on pages 4-4 to 4-8 of the May 2013 document: Suplee, Michael W., V. Watson, A. Varghese, and Joshua Cleland. 2008. *Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers*. Helena, MT: MDEQ Water Quality Planning Bureau.

The EPA examined Montana's process for deriving reach-specific criteria and finds the criteria, reflecting natural background conditions, are scientifically defensible and protective of the aquatic life designated use. These provisions are approved.

DURATION AND FREQUENCY

For all NNC adopted by Montana for wadeable streams and segments of the Yellowstone River, Department Circular DEQ-12A defines the duration and frequency associated with the standard as: "The average concentration during a period when the standards apply may not exceed the standards more than once in any five-year period, on average." (Section 3.0, Endnote 4). This duration and frequency means that, for a given waterbody, the TN and TP concentrations must not exceed the applicable criterion concentration more than once in a 5-year period.

Montana's determined the once in 5-year recurrence frequency based on an analysis of a long-term dataset (1998-2009) from the Clark Fork River where NNC have been approved by the EPA since 2003. The state's analysis examined TN and TP data from sites along the Clark Fork River[27] that were meeting and exceeding the numeric chlorophyll criterion. Results of that analysis showed that: *"Sites that experience greater than about 25-30% exceedance of the nutrient standards will develop nuisance benthic algal growth, i.e., growth equal to or greater than 150 mg Chl a/m².* " The state used this information to inform their selection of the one in 5-year recurrence frequency since that frequency is similar to a 20% exceedance rate. Montana also noted that a once in 5-year recurrence frequency is more protective than the EPA's long-standing recommendation (i.e., once in three years).

Basis for Approval

The EPA determined that such a frequency of exceedances would still protect the designated use because it would allow water bodies enough time to recover from occasionally elevated levels of nitrogen and phosphorus concentrations. The EPA has concluded that the adopted duration and frequency provisions are consistent with 40 CFR § 131.11(a)(1) of EPA's water quality standards regulation. Accordingly, the EPA is approving these provisions.

ASSESSMENT METHODOLOGY

Montana's current assessment methodology for nutrients is based on the existing narrative standard. The EPA recognizes and supports the state's decision to apply the draft NNC as part of a weight-of-evidence approach to interpret the narrative when developing its 303(d) list. Now that the state has adopted NNC applicable to certain waters and waterbody types and the EPA has approved such standards as discussed above, the EPA fully expects Montana to revise and update its nutrient assessment methodology to be consistent with the newly adopted and EPA-approved NNC. These revisions should be completed prior to the 2016 Integrated Reporting cycle to ensure that nutrient-related attainment decisions reflect compliance with the newly adopted and EPA-approved numeric criteria values.[28]

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[27] See pages A8-A14. Suplee, M.W., and R. Sada de Suplee, 2011 Assessment Methodology for Determining Wadeable Stream Impairment Due to Excess Nitrogen and Phosphorus Levels. Helena, MT: Montana Dept. of Environmental Quality.

[28] For impairment decisions and total maximum daily loads (TMDLs), CWA § 303(d)(1)(A) requires that each State shall identify "those waters within its boundaries for which the effluent limitations required by section 301(b)(1)(A) and section 301(b)(1)(b) are not stringent enough to implement any water quality standard applicable to such waters" (emphasis added). Accordingly, listing decisions must consider the underlying designated use and criteria.

DOWNSTREAM USE PROTECTION

Protection of downstream waters is required by language included in Endnotes 2 in Department Circular DEQ-12A Section 3.0:

> (2) Within and among the geographic regions or watersheds listed, base numeric nutrient standards of the downstream reaches or other downstream waterbodies must continue to be maintained. Where possible, modeling methods will be utilized to determine the limitations required which provide for the attainment and maintenance of water quality standards of downstream waterbodies.

Basis for Approval

Montana's downstream provision provides a process that will serve to ensure that water quality standards are maintained both near and far-field. Montana's provision is consistent with both EPA's regulation at 40 CFR § 131.10(b) and the following EPA recommended language for developing a narrative downstream protection criterion:[29]

> "All waters shall maintain a level of water quality that is demonstrated by water quality modeling to provide for the attainment and maintenance of the water quality standards of downstream waters, including the waters of another state."

Since Montana is not adopting NNC for any downstream waterbodies such as lakes or reservoirs at this time, the EPA concludes the state's decision to adopt a narrative downstream provision is appropriate. In cases where a downstream water quality standard is not attained, the EPA's expectation is that Montana would evaluate the upstream waterbody(ies), based on the narrative downstream criterion, to determine impairment under CWA Section 303(d).

This provision is approved.

PERMITTING COMPONENTS (DEQ-12A SECTION 2.1)

Section 2.1 of DEQ-12A identifies the required reporting limits for calculating total nutrient concentrations for TN and TP. The EPA is not acting on the reporting requirements today because the EPA determined they are not water quality standards requiring Agency review and approval under CWA § 303(c).

VARIANCE AUTHORIZATION PROVISIONS

Section ARM 17.30.660(1) authorizes the general and individual variances for nutrients once the BER adopts the NNC.

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[29] Templates for Narrative Downstream Protection Criteria in State Water Quality Standards: http://water.epa.gov/scitech/swguidance/standards/narrative.cfm

Basis for Approval

The EPA has reviewed this provision and determined that it is consistent with the EPA's requirements. The EPA's water quality standards regulation (40 CFR § 131.13) provides that variance policies may be adopted at state discretion, and that such general policies are subject to review and approval by the EPA.[30,31] The EPA approves ARM 17.30.660(1).

GENERAL VARIANCES FOR PUBLIC AND PRIVATE DISCHARGERS

A variance is a "time-limited designated use and criterion that is targeted to a specific pollutant(s), source(s), and/or water body or waterbody segment(s) that reflects the highest attainable condition during the specified time period."[32] The EPA encourages states and authorized tribes to utilize WQS variances, where appropriate, as an important WQS tool that provides time to make progress towards attaining the underlying designated use and criteria. The EPA has offered its position and guidance relating to variances through Office of General Counsel legal decisions,[33] guidance, memoranda, and approval actions for many years.[34]

The EPA's position is that it could approve a variance for a specific discharger or group of dischargers where the state satisfies the requirements in 40 CFR Part 131 for removing a designated use.[35] As such, the state must demonstrate that it is not feasible for the discharger or group of dischargers to attain the WQBEL(s) derived from the applicable designated use and criteria during the term of the variance due

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[30] On September 4, 2013 the Agency proposed revisions to its WQS regulation that include new requirements addressing WQS variances. The comment period on the proposed rule closed on January 2, 2014.

[31] Guidance regarding State options is provided in Section 5.3 of the EPA Water Quality Standards Handbook (EPA-823-B-94-005, August 1994). http://water.epa.gov/scitech/swguidance/standards/handbook/index.cfm.

[32] Water Quality Standards Regulatory Clarifications, 78 Fed. Reg. 54517, 54531 (September 4, 2013).

[33] It has been EPA's position since 1977 that, where a state satisfies all of the requirements in 40 CFR Part 131 for removing designated uses (or subcategories of uses), EPA could also approve a state decision to limit the applicability of the use removal to only a single discharger and/or a single criterion via a variance for a limited time period, while continuing to apply the underlying use designation and criteria to the waterbody as a whole (i.e., the underlying use designation and criteria would apply to all other dischargers other than the one for which a variance has been granted). This position was set forth in a Decision of the EPA General Counsel (In Re Bethlehem Steel Corporation, No. 58, March 29, 1977). The General Counsel's decision reasoned that such a state decision can be approved by EPA as being consistent with the CWA and 40 CFR Part 131 because the state's action in limiting the applicability of an otherwise approvable use removal to a single discharger and a single criterion for a limited time period would be more stringent than if the state made the use removal applicable to the water body as a whole; and Section 510 of the CWA allows states to adopt standards more stringent than necessary to meet the CWA's requirements. See 58 Fed. Reg. 20802, 20921-22 (April 16, 1993).

[34] The EPA's memoranda discussing variances are available on the EPA's website at http://water.epa.gov/scitech/swguidance/standards/handbook/chapter05.cfm or http://water.epa.gov/scitech/swguidance/standards/library/index.cfm.
http://water.epa.gov/scitech/swguidance/standards/upload/2008_08_04_standards_wqsvariance.pdf.

[35] EPA has explained a state or authorized tribe may streamline its variance process by granting one variance that applies to all these dischargers (i.e., a multiple discharger variance) where the state or authorized tribe can demonstrate that that the designated use and criterion is unattainable as it applies to multiple permittees because they are all experiencing challenges in meeting their WQBELs for the same pollutant for the same reason, regardless of whether or not they are located on the same water body, so long as the variance is consistent with the CWA and EPA's implementing regulations. *See* Water Quality Standards Regulatory Clarifications, 78 Fed. Reg. 54517, 54531-32 (September 4, 2013) and EPA's FAQs on multiple discharger variances available at: ahttp://water.epa.gov/scitech/swguidance/standards/upload/Discharger-specific-Variances-on-a-Broader-Scale-Developing-Credible-Rationales-for-Variances-that-Apply-to-Multiple-Dischargers-Frequently-Asked-Questions.pdf.

to at least one of the factors listed in 131.10(g).[36] Section 131.10(g) includes the following factors: (1) naturally occurring pollutant concentrations prevent the attainment of the use; (2) natural, ephemeral, intermittent or low flow conditions or water levels prevent the attainment of the use, unless these conditions may be compensated for by the discharge of sufficient volume of effluent discharges without violating state water conservation requirements to enable uses to be met; (3) human caused conditions or sources of pollution prevent the attainment of the use and cannot be remedied or would cause more environmental damage to correct than to leave in place; (4) dams, diversions, or other types of hydrologic modifications preclude the attainment of the use, and it is not feasible to resort the water body to its original condition or to operate such modification in a way that would result in the attainment of the use; (5) physical conditions related to natural features of the water body such as lack of a proper substrate, cover, flow, depth, pools riffles, and the like, unrelated to water quality, preclude attainment of aquatic life protection uses; or (6) controls more stringent than those required by sections 301(b) and 306 of the Act would result in substantial and widespread economic and social impact.

The EPA reviewed Montana's basis[37,38] for determining that it is reasonable to grant multiple public and multiple private dischargers throughout the state with general variances of up to 20 years based on a demonstration that it is infeasible to meet water quality-based effluent limits based on the NNC (and by extension infeasible to attain the designated use for that limited time) "end-of-pipe" because meeting such limits would cause substantial and widespread economic and social impacts (see 40 CFR § 131.10(g)(6)) on a statewide basis. This analysis is the focus of the EPA review discussed below.

Economic Analysis for POTWs

For the economic analysis of publicly-owned wastewater treatment plants (POTWs) within the state, Montana referred to the EPA's 1995 economic guidance to evaluate substantial and widespread economic impacts.[39] Montana identified the 107 actively discharging POTWs within the state, and completed the analysis of economic impacts for 24 of the 107 dischargers across Montana. The state considered this subset to be a representative subsample of the economic and technological conditions for the entire population of dischargers. The state's analysis examined effluent data and financial information for all 12 POTWs that discharge more than 1 million gallons per day (MGD); four of the 12 facilities that discharge less than 1 MGD; and eight of the 83 lagoon systems.[40] Appendix A of the state's economic demonstration[41] includes the detailed cost analyses for each plant.

Using EPA's guidance as a starting point for its analysis, the state applied three "tests" to determine if the cost to meet the NNC would cause substantial economic and social impacts for the community: 1)

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[36] Id.

[37] Blend, Jeff; Suplee, Michael. 2011. Demonstration of Substantial and Widespread Economic Impacts to Montana That Would Result if Base Numeric Nutrient Standards had to be Met in 2011/2012. Helena, MT: Montana Dept. of Environmental Quality.

[38] Blend, Jeff; Suplee, Michael. 2012. Demonstration of Substantial and Widespread Economic Impacts to Montana That Would Result if Base Numeric Nutrient Standards had to be Met by Entities in the Private Sector in 2011/2012. Helena, MT: Montana Dept. of Environmental Quality.

[39] U.S. Environmental Protection. 1995. Interim Economic Guidance Workbook. Washington, DC: U.S. Environmental Protection. Report EPA-823-B-95-002.

[40] Lagoons refer as "facultative waste stabilization ponds" (USEPA 2002. Wastewater Technology Fact Sheet) http://water.epa.gov/scitech/wastetech/upload/2002_10_15_mtb_faclagon.pdf. In Montana, this includes aerated and non-aerated facultative waste stabilization ponds.

[41] Blend, Jeff; Suplee, Michael. 2011. Demonstration of Substantial and Widespread Economic Impacts to Montana That Would Result if Base Numeric Nutrient Standards had to be Met in 2011/2012. Helena, MT: Montana Dept. of Environmental Quality.

Municipal Preliminary Screener (MPS) test; 2) secondary score; and 3) the widespread test. The MPS and secondary score constitute an evaluation of whether the population that is expected to bear the cost will incur "substantial" economic impacts due to the implementation of the pollution control costs. The MPS "screener" test establishes whether a community can clearly pay for the project without incurring any substantial impacts. If a community did not pass the "screener" test, the state used the secondary test to incorporate a characterization of the community's current financial and economic well-being. Together these two tests can demonstrate whether or not a community has "substantial" economic impacts. In order to derive the MPS, the state needed to estimate the compliance costs to meet the NNC. The state first described the current treatment technology and nutrient effluent concentrations for each of the 24 facilities. Next, the state identified additional treatment technology needed to achieve the NNC after examining a variety of different treatment processes. Effluent concentrations associated with enhanced biological nutrient removal technology are the best currently being achieved anywhere in the U.S. at full-scale wastewater treatment facilities. According to Montana's analysis, effluent concentrations using enhanced biological nutrient removal (EBNR) technology ranged from 3000 - 4000 μg/L TN and 50 – 70 μ/L TP.[42,43] If those concentrations were end-of-pipe (no mixing zone) limits, they would not meet the nitrogen criteria (see Table 12A-1 on page 5) and would not necessarily meet the phosphorus criteria (see Table 12A-1 on page 5). Therefore, Montana did not use EBNR as the basis for determining compliance costs.

Instead, the state considered reverse osmosis (RO) to be the most advanced treatment method with the greatest likelihood of achieving Montana's NNC, which includes nitrogen and phosphorus criteria. Wastewater engineering reports document that RO can achieve concentrations of less than 2000 μg/L TN and may meet concentrations of 1000 μg/L TN (depending on a number of factors) and less than 0.010 μg/L TP.[44] Based on this information, Montana determined that RO was the only available technology for facilities to implement in order to meet WQBELs derived to meet the state's dual NNC.

Montana calculated the cost of compliance based on RO using data available from the Interim Water Environment Research Foundation (WERF) study.[45] The WERF study identifies different treatment levels and their associated capital and operations costs. To calculate the total annual pollution control costs for each facility, current effluent concentrations were compared to the costs of treating 50% and 100%, of the plant's effluent using RO. Both scenarios were run because meeting the NNC may require reducing influent TN concentrations by using a two-pass RO system (i.e., treating 100%),[46] Montana next calculated the total annual pollution control cost per household, including the cost of the project and existing pollution control costs.

Montana also completed an overall sensitivity analysis to derive the MPS value. In the sensitivity analysis, the state examined the effect of different discount rates (i.e., using 7% instead of 5%); labor costs (labor was excluded from the WERF cost estimates); and treating 100% of the effluent using RO.

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[42] Hartman, Pamela, and J. Cleland. 2007. Wastewater Treatment Performance and Cost Data to Support and Affordability Analysis for Water Quality Standards.

[43] Presentation by Dave Clark, HDR Consulting to MDEQ Nutrient Workgroup. Achievable Technology for Municipal Wastewater Systems. 09/17/2009.

[44] Falk, M. W., J. B. Neethling, and D. J. Reardon. 2012. Striking the Balance Between Nutrient Removal in Wastewater Treatment and Sustainability. IWA Publishing. U.S. Environmental Protection. 1995. Interim Economic Guidance Workbook. Washington, DC: U.S. Environmental Protection. Report EPA-823-B-95-002.

[45] Id.

[46] See page 18 of MDEQ's economic demonstration for more detail. Blend, Jeff; Suplee, Michael. 2011. Demonstration of Substantial and Widespread Economic Impacts to Montana That Would Result if Base Numeric Nutrient Standards had to be Met in 2011/2012. Helena, MT: Montana Dept. of Environmental Quality.

The EPA found the sensitivity analysis to represent the range of circumstances that could be encountered.

The MPS value represents the cost of annualized proposed pollution controls per household. The EPA's economic guidance states that MPS values greater than 2% indicate that the project may place an unreasonable financial burden on many of the households within the community. If the MPS suggests substantial impacts may be possible (i.e., >1%) or more likely (i.e., >2 %), the EPA guidance recommends performing the secondary test to confirm substantial economic and social impact. Secondary scores describe the socioeconomic health of the community in more detail and demonstrate the community's ability to obtain financing for wastewater improvements. In its approach, Montana chose to use its own updated list of indicators to determine the secondary score.[47, 48] Using the data for its updated list of indicators, Montana calculated the secondary scores for the 24 communities. Montana then used secondary scores in combination with the MPS results and the sensitivity analysis to determine whether implementing the pollution control costs would cause "substantial" economic impacts to the community. The state asserted that based on the results of the secondary scores and the MPS values, all 107 communities showed substantial economic impacts.

Lastly, the state evaluated statewide economic impacts of meeting the NNC through application of the "widespread" test. The "widespread" test examines the impacts to the larger affected community, recognizing that the financial impacts associated with the discharger treating to the NNC could cause "far reaching and serious impacts to the community".[49] Montana described the potential cumulative adverse economic impacts that could occur including: a) the expense associated with replacing lagoons with mechanical treatment plants for the majority of communities; b) the state's current ranking as 41st in the nation in per capita income; c) impacts to struggling small towns lacking diversified economies; d) challenges with finding qualified wastewater treatment plant operators; and e) impacts to other community infrastructure needs. In addition, the state described the environmental consequences associated with building RO treatment systems (e.g., brine disposal, increased greenhouse gas emissions). The state concluded that Montana would experience widespread economic impacts if communities were required to implement the necessary pollution control costs without the added flexibility of staging attainment by dischargers over up to 20 years.

Economic Analysis for Private Facilities

Montana's showing of economic impacts to private-sector dischargers was modeled on the EPA's economic guidance and is similar to the public sector analysis.[50] First, the state identified 51 private dischargers from a variety of sectors (e.g., metal mining; coal mining; oil and gas development; oil and gas refineries; etc.) that may be affected by adoption of NNC. NPDES water discharge permits, monitoring data, and the statement of basis for these dischargers were examined to evaluate current treatment levels for each facility. The state's analysis assumed that the costs of compliance would be incurred by the businesses and not transferred to Montana households. Similar to the public sector analysis, Montana projected the costs of achieving the NNC based on the following assumptions: a) treatment of 50% and 100% of the facility's effluent using RO would be required; b) discount rates

[47] See Appendix C of Blend and Suplee (2011). Demonstration of Substantial and Widespread Economic Impacts to Montana That Would Result if Base Numeric Nutrient Standards had to be Met in 2011/2012.
[48] Memo submitted to the EPA from Jeff Blend, MDEQ, on 12/09/2014. Changes to the Individual Variance Made by the NCAAG (Nutrient Criteria Affordability Advisory Group).
[49] U.S. Environmental Protection. 1995. Interim Economic Guidance Workbook. Washington, DC: U.S. Environmental Protection. Report EPA-823-B-95-002.
[50] Id.

would be 5% or 7%; and c) labor costs may vary from 15% to 48%. The state's private sector economic analysis also included a sensitivity analysis. Where possible, plant level information was used to determine current and projected costs of meeting the NNC.

The EPA guidance does not identify a specific economic hardship threshold (i.e., 2% MPS for the public sector) that can be applied to determine whether private-sector economic impacts are substantial. Therefore, the state examined economic impacts to individual facilities and also at a statewide scale. Montana presented financial analyses completed for several of the larger businesses as a signal of the economic impacts that could also occur to smaller businesses if facilities were required to treat to the NNC.[51] This review suggested larger plants may experience impacts such as a loss in revenues; layoffs; or scaling back production. In some cases, plants may have to shut down, affecting the financial status of the broader community.

Montana also evaluated sector-level estimates associated with meeting NNC. Montana's analysis estimated the amount of total annual revenue that businesses would spend to meet the NNC. Additionally, Montana's private-sector economic demonstration includes several case studies of individual businesses working to implement rigorous nutrient controls. These case studies offer insights into the implications of meeting the adopted NNC for private businesses-- documenting the technological and financial barriers that may be encountered.

The state's economic analysis concludes with the "widespread" test which discusses the projected statewide implications to private businesses including: a) recent impacts from the recession; b) companies deciding not to locate in Montana to avoid costs associated with meeting the NNC without the possibility of staging attainment by dischargers over up to 20 years; and c) impacts of business closures including loss of higher wage paying jobs on the local and statewide economy. As noted above, based on the EPA's review of the available treatment technologies for total nitrogen, there is not an existing technology currently available that would reliably meet Montana's dual NNC, especially stringent nitrogen criteria (1300 µg/L TN (warm water); 300 µg/L TN (cold water)). This presents similar difficulties for some industrial dischargers who, without available treatment, could be in the position of halting operations entirely in the state. Closure of these facilities could result in significant job losses in the Montana.

Basis for Approval

In the EPA's review of Montana's economic demonstration, the EPA first reviewed the list of dischargers included in the state's analysis. The EPA notes that an estimated thirty dischargers included in the state's economic analysis discharge into non-wadeable rivers for which numeric nutrient criteria have not yet been derived or adopted. Based on ARM 17.30.660(1), the EPA understands that these facilities will continue to be subject to Montana's existing narrative criterion instead of the NNC and therefore the EPA's approval of general variances today does not include these dischargers. Additionally, the state's economic analysis included dischargers currently covered by a general permit for domestic sewage lagoons. The EPA's approval of general variances today does not apply to these lagoons because they are not yet subject to the NNC.

The EPA evaluated whether including these facilities in the state's economic analysis affected the final

[51] See Table 5; Pages 8-10. Blend, Jeff; Suplee, Michael. 2012. Demonstration of Substantial and Widespread Economic Impacts to Montana That Would Result if Base Numeric Nutrient Standards had to be Met by Entities in the Private Sector in 2011/2012. Helena, MT: Montana Dept. of Environmental Quality.

outcome. From the EPA review, it appears that facilities discharging into non-wadeable rivers without established NNC are similar in composition to the subset of facilities with established NNC that were used in the state's economic analysis. For example, for the public sector, facilities on non-wadeable rivers ranged from larger more affluent communities with mechanical plants (i.e., Billings, Livingston) to small towns with lagoon systems. For the private sector, facilities discharging into non-wadeable rivers includes a mix of larger, multi-national private dischargers with greater financial capabilities to make capital improvements (i.e., Exxon, Conoco) to facilities that may not be currently discharging. By including both highly profitable and potentially nondischarging facilities in their economic analysis, it is possible the state's economic analysis may have underestimated the economic impacts associated with meeting the NNC. The EPA concludes that including these facilities from the economic analysis does not undermine the final conclusion in the state's economic analysis that meeting the NNC would result in substantial and widespread economic and social impacts for all dischargers subject to the NNC.

For the public sector economic demonstration, the EPA reviewed the list of public dischargers included in the state's analysis. The state's economic analysis focused on those communities with the highest likelihood of being able to afford to meet the NNC. By demonstrating that the largest, and generally most affluent, communities with already-sophisticated systems in place (e.g., biological nutrient removal) and/or that large populations where additional costs could be dispersed (i.e., economies of scale) would face economic hardship, Montana demonstrated that the remaining dischargers (primarily lagoons) would also face economic hardship if required to meet the NNC. These dischargers would have to absorb much higher costs of additional technology (e.g., RO plant) with less population to absorb the costs. Assuming these remaining dischargers have at most the same median household income as the other communities, the net effect is a higher MPS value. Since the subset of communities examined in Montana's analysis exceeded the 2% threshold, Montana concluded the remaining dischargers would also have MPS values above the 2% threshold. The EPA finds this assumption reasonable.

The EPA also evaluated the state's assumption that facilities would need to meet the NNC at the end-of-pipe. There were several factors relevant to determining whether a facility would need to meet the NNC at the end of end-of-pipe including: whether the facility discharges into a waterbody on the state's 303(d) list as impaired for nutrients; whether any mixing zone is available; and whether the facility discharges into an intermittent waterbody or waterbody where the 14Q5 would likely be zero. The EPA concludes that the state's assumption that criteria would need to be met at the end-of-pipe is reasonable.

Next, the EPA examined Montana's assumption that RO would be required to meet the NNC by reviewing the available literature on treatment technologies; identifying the effluent concentrations that can reliably be achieved; and consulting with wastewater experts both within the EPA as well as outside of the Agency. The EPA recognizes that treatment technologies other than RO may meet some of Montana's numeric TP criteria if it was the only criteria that Montana had adopted and dischargers were treating only for total phosphorus.[52, 53,54] For example, case studies from Colorado (Cherry Creek Reservoir Control Regulation), Utah (Snyderville Water Reclamation Facility)[55] and Montana

[52] Water Environment Research Federation. 2010. Nutrient Management: Regulatory Approaches to Protect Water Quality. Volume 1 – Review of Existing Practices.
[53] EPA. 2009. Nutrient Control Design Manual. State of Technology Review Report. EPA/600/R-09/012.
[54] EPA. 2010. Nutrient Control Design Manual. EPA/600/R-10/100.
[55] Pers. Com. February 9, 2015.

(Kalispell)[56] demonstrate that, while expensive, dischargers can use chemical addition and/or microfiltration to consistently achieve total phosphorus concentrations of 0.050 µg/L. However, chemical addition or microfiltration cannot achieve the nitrogen criteria component of Montana's NNC. Montana's approach to addressing nutrient pollution is based on the need for managing both total nitrogen and total phosphorus in order to manage the full nutrient pollution problem, which the EPA supports.[57] In the scientific justification for adopting an NNC that necessarily includes both TN and TP criteria, Montana states:

> The Department is recommending both TN and TP criteria for stream protection. Phosphorus (P) control is sometimes promoted as the only approach needed to limit eutrophication, this being based largely on the more economical removal of P from wastewater and the assumption that P can be made to become limiting in the waterbody. But data pertaining to streams and rivers indicate that it would be unwise to adopt only P criteria. Mixed assemblages of benthic algae are very often limited by nitrogen or nitrogen and phosphorus (co-limitation) in the region's flowing waters. A P-only approach, in order to work, would require that P standards be set to the very low background levels observed in our western region's reference sites (e.g., 10 µg TP/L). If the P standard were not set to natural background, and no controls on N were undertaken, then the commonly occurring N limitation or N and P co-limitation would lead to algal growth stimulation nonetheless. Worse yet, in the long term, a P-only strategy would result in highly skewed (elevated) N:P ratios accompanying the low P levels. These management-induced conditions might control green algae biomass but may lead to nuisance blooms of the diatom algae *Didymosphenia geminata,* which has in recent years formed nuisance blooms in rivers and streams in Montana and world-wide. (Executive Summary). [58]

Determining the cost of compliance with Montana's NNC requires identification of treatment technologies that will meet both the TN and TP criteria. Treatment options that meet one criteria but not the other would not ensure protection of the aquatic life designated use.

Based on the EPA's review of the available treatment technologies for total nitrogen, there is not an existing technology currently available that would reliably meet Montana's stringent NNC which includes both nitrogen and phosphorus criteria. RO is the only treatment option that has the potential to remove the total nitrogen component of the NNC to concentrations of approximately 1000 µg/L TN. Case studies examining RO performance indicate that the reliability and consistency of meeting a TN concentration of 1000 µg/L TN are highly variable and depend on the TN concentrations of the influent, total dissolved solids concentrations, temperature and pH. Removal of refractory dissolved organic nitrogen has also been shown to be a challenge when striving to meet such a low concentration.[59] Therefore, using a single-pass RO system to meet a 1300 µg/L TN monthly summer average criterion for warm water streams is considered unreliable. Because there are no existing treatment technologies that can reliably achieve the nitrogen criteria of the NNC for wadeable streams, the EPA supports Montana's view that achieving WQBELs based on the NNC and thus attaining the NNC (and the designated use) is infeasible until treatment methods improve or ambient levels of nutrients in the streams decrease to the point that effluent discharge concentrations do not need to be equal to the NNC,

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[56] EPA. 2008. Municipal Nutrient Removal Technologies Reference Document. Volume II – Appendices. EPA-832-R-08-006. http://water.epa.gov/scitech/wastetech/upload/2008_10_06_mtb_mnrt-volume2.pdf

[57] EPA. 2012. http://www2.epa.gov/sites/production/files/documents/nandpfactsheet.pdf

[58] Suplee, M.W1., and V. Watson2, 2013. Scientific and Technical Basis of the Numeric Nutrient Criteria for Montana's Wadeable Streams and Rivers—Update 1. Helena, MT: Montana Dept. of Environmental Quality.

[59] Falk, M. W., J. B. Neethling, and D. J. Reardon. 2012. Striking the Balance Between Nutrient Removal in Wastewater Treatment and Sustainability. IWA Publishing.

otherwise substantial and widespread economic and social impacts will occur. Optimization studies, including efficiencies that could be obtained through trading with nonpoint sources, may illuminate such opportunities.

For the public sector dischargers, the EPA concludes that based on the above, requiring public sector dischargers to meet WQBELs based on Montana's adopted NNC would result in substantial and widespread economic and social impacts for all POTWs covered by a general variance. The state's analysis meets the requirements of 40 CFR § 131.10(g) and justifies a variance of up to 20 years for POTWs.

For the private sector economic demonstration, the EPA concludes that the state's submission meets the requirements of 40 CFR § 131.10(g) and justifies a variance of up to 20 years by demonstrating that requiring private sector facilities to meet WQBELs during the period of the variance based on Montana's adopted NNC would result in substantial and widespread economic and social impact. Given that there is no feasible technology to reliably meet the TN criteria, a broad spectrum of facilities and industries would be forced to substantially alter or halt operations. The resulting cascade of impacts would be felt throughout all communities statewide. Montana's variance provisions provide needed time to determine how to achieve compliance with necessary effluent limits based on the NNC, and ensure that progress toward that goal will proceed in a timely manner.

If at the time of permitting, Montana determines that, based on site-specific facts and details (e.g., dilution, alternatives to discharge, installing less expensive treatment technology), an individual discharger can meet the NNC-based limits, then the discharge permit would include such limits. Where necessary and appropriate, a compliance schedule may be included in the permit. This approach is consistent with Montana's regulatory language that variances may be provided for up to 20 years, or for a shorter duration, should the state determine that is appropriate. Another option would be for the discharger to apply for an individual variance based on a site-specific demonstration that the discharger cannot afford to meet such NNC-based limits.

General Variance Considerations and Water Quality Protections that Apply While the Variance is in Effect

ARM 17.30.660(2) establishes that any discharger covered by a general variance must meet the requirements described in DEQ-12B. This provision documents that "the decision to grant the general variance must be reflected in the permit that is made available for public comment." Section 2.0 of DEQ-12B provides additional detail regarding implementation, stating that general variance coverage will be implemented through the permitting process and that permits will include the period of the variance and the interim requirements for each discharger covered under a general variance.

Section 2.0 of DEQ-12B provides additional detail regarding general variances including: a) interim end-of-pipe treatment requirements which expire on July 1, 2017 ; b) the maximum 20 year duration of a variance; c) permitting details associated with the variance; and d) review requirements of the justification for the variance and future end-of-pipe treatment requirements to make progress towards the NNC.

This section goes on further to define the end-of-pipe interim treatment requirements at Table 12B-1 (see figure) for three categories of dischargers: 1) facilities with discharge volumes greater than 1 MGD; 2) dischargers with volumes less than 1 MGD; and 3) lagoons. The interim treatment requirements shall

be applied as a monthly average as defined in Sections 1.1 and 2.1 of DEQ-12B.

Section 2.0 of DEQ-12B requires that, after June 1st, 2016, and triennially thereafter, Montana review the economic justification for the general variances as well as the cost and effluent concentrations associated with available treatment technologies. Findings from this review will determine the next set of interim limits that apply under the general variances after 2017. The state will solicit public comment on its draft findings and will initiate rulemaking if there is a need to revise the interim limits and/or continue the general variance without modifications. Results of the rulemaking will be submitted to the EPA for review and approval.

Table 12B-1. General variance end-of-pipe treatment requirements.

	Monthly Average	
Discharger Category	**Total P (µg/L)**	**Total N (µg/L)**
> 1.0 million gallons per day	1,000	10,000
< 1.0 million gallons per day	2,000	15,000
Lagoons not designed to actively remove nutrients	Maintain current performance	Maintain current performance

Section 2.0 clarifies that permit limits implementing the end-of-pipe treatment requirements and NNC will be expressed in loads. The rule language also indicates that compliance schedules can be incorporated into the permit to allow time to meet the interim treatment requirements.

Section 2.1 of DEQ-12B requires permittees covered by a general variance to complete an optimization study within two years of receiving the variance. The optimization study must explore alternatives to reduce nutrient loading such as nutrient trading, facility optimization without substantial investment in new infrastructure, reuse, recharge, and land application.

Basis for Approval

The EPA finds Montana's general variances for public and private dischargers to be reasonable and consistent with CWA requirements. As discussed above, the state's economic analyses demonstrate that the facilities subject to WQBELs based on the NNC need a variance because meeting WQBELs based on the NNC during the term of the variance would cause substantial and widespread economic and social impact, consistent with 40 CFR § 131.10(g)(6). In addition, the maximum 20-year time frame of the general variances combined with the requirement for the state to review every three years both the justification for the general variances and to review, obtain public input and adopt new interim treatment requirements provides assurance that these dischargers will be expected to achieve specific numeric interim treatment requirements throughout the variances in order to make progress towards achieving

the target effluent limitations based on the underlying NNC. Montana documented the rationale for the maximum 20-year variance limit in DEQ-12B (General Introduction) stating:

> Because many of the base numeric nutrient standards are stringent and may be difficult for MPDES permit holders to meet in the short term, Montana's Legislature adopted laws (e.g. §75-5-313, MCA) allowing for the achievement of the standards over time via the variance procedures found here in Circular DEQ-12B. This approach should allow time for nitrogen and phosphorus removal technologies to improve and become less costly, and to allow time for nonpoint sources of nitrogen and phosphorus pollution to be better addressed." (underline added)

Montana's approach facilitates long-term facility planning by defining the NNC as the highest attainable condition (HAC) for its waters and establishing a maximum of 20 years to achieve that HAC. Given the current lack of existing treatment technologies that can reliably achieve effluent limits based on the NNC, specifically the stringent nitrogen criteria, discussed above, the variance process provides time for dischargers to identify and implement the most cost effective method for making progress towards meeting the NNC while also ensuring that the NNC remains the goal. Montana's nutrient rules establish the NNC as the long-term HAC with interim milestones (i.e., interim treatment requirements) required for dischargers to meet in the near term:

> "Variances from the standards may be granted for up to 20 years. Thus, 75-5-313,MCA, allows for the base numeric nutrient standards to be met in a staged manner over time, as alternative effluent management methods are considered, nutrient removal technologies becomes more cost-effective and efficient, and nonpoint sources of nutrients are addressed." (Statement of Reasonable Necessity ARM 17.30.660)

To ensure that dischargers are making meaningful progress toward the HAC throughout the duration of the variance, Montana's approach incorporates short-term interim milestones, adopted on a triennial basis. The first set of milestones are the end-of-pipe treatment requirements established by the MT statute and re-iterated in Table 12B-1 that expire on July 1, 2017, after which Montana will go through a public rulemaking process to establish the next set of interim treatment requirements. The procedure established in Montana's regulations provides accountability that dischargers will make progress towards meeting the NNC by the end of the general variance provided that the triennial review process is implemented appropriately and effectively. This process should ensure that the water quality protection requirements imposed by the variances keep pace with what is feasible to achieve. This approach also affords the public an opportunity to review and comment on the proposed milestones. Montana will submit a new WQS rule package including the interim milestones applicable for the next three-year period to the EPA for review and approval.

Based on prior conversations with the state, the EPA understands that Montana will include limits based on the NNC in the permit fact sheet. The EPA supports and encourages this practice so that dischargers are fully aware of what will be expected of them at the end of the variance period.

Montana's approach is comprehensive and provides time for dischargers to incrementally work to achieve stringent WQBELs based on the protective NNC. The EPA supports Montana's decision to establish interim treatment requirements and to require a review of the interim treatment requirements and underlying variance justifications on a triennial basis. Not only will Montana's rules as a whole ensure that dischargers are making progress towards achieving the HAC in a process that includes public

input and oversight by the EPA, but this approach also provides incentives to maximize optimization, develop innovative treatment technologies, and look toward nonpoint source reductions, especially for nitrogen, to facilitate that the NNC will be achieved in 20 years.

The initial set of end-of-pipe treatment requirements included in the rulemaking expire on June 1, 2017. This expiration is appropriate given that the state statute authorizing the general variances, MCA 75-5-313, sets forth that particular set of treatment requirements for only that time frame. As the expiration date approaches for the initial set of treatment requirements, the EPA fully expects Montana to adopt the next set of general variance milestones that will ensure dischargers continue to reduce nutrient loads and will ensure Montana is on a pathway to protect aquatic life designated use and attain the NNC. Such interim requirements should, themselves, reflect the best that dischargers can achieve in that time period and be based on 1) information collected during the optimization studies completed during the first phase of the general variances; and 2) additional analyses about what is affordable for facilities under the substantial and widespread economic and social test.

Montana's nutrient rules (specifically section 2.1 of DEQ-12B) define the expectations for the optimization studies.[60] Optimization is a tool that, when effectively implemented and sustained, can achieve remarkable nutrient reductions at much lower costs and within much shorter timeframes (~3 years).[61] Optimization case studies demonstrate that plant performance can be improved to achieve TN and TP concentrations below 10,000 µg/L and 1000 µg/L respectively.[62] Optimization work recently completed at several Montana wastewater treatment plants (e.g., Manhattan, Chinook, Conrad) demonstrate improved plant performance with effluent total nitrogen concentrations reduced by as much as 50%.[63,64] By coupling the interim treatment requirements with an optimization requirement, Montana's approach facilitates shorter-term nutrient reductions from dischargers that will inform future interim treatment requirements.

Section 2.0 includes rule language that a compliance schedule may be incorporated into a permit to allow time to meet the interim treatment requirements. Such schedules are appropriate where compliance with the WQBEL is feasible but time is needed. For example, facilities may need time to secure funding[65], install treatment technology and implement the steps necessary to meet the WQBEL. The duration of a compliance schedule is determined based on discharger-specific information and must ensure compliance as soon as possible and be consistent with EPA's federal regulations at 40 CFR § 122.47. The state's decision to authorize permit compliance schedules for purposes of implementing such limits is fully consistent with the state's more general authority[66] to establish permit compliance schedules for <u>any</u> water quality-based effluent limit.

Based on our review, the EPA concludes that ARM Section 17.30.660(2),Sections 2.0 and Section 2.1 of DEQ-12B implementing general variances for both public and private dischargers are consistent with the EPA's regulations and are approved.

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[60] Section 2.1 of Circular DEQ-12B requires permittees covered under a general variance to complete an optimization study.
[61] Wastewater Nitrogen & Phosphorus Removal without Plant Upgrades: Optimizing the Operation of Existing Facilities. The Water Planet Company. 10 December 2013 Presentation to EPA Region 8.
[62] <u>Id</u>.
[63] Paul LaVigne, Montana Department of Environmental Quality, personal communication, March 24, 2014.
[64] Grant Weaver. The Water Planet Company. http://www.cleanwaterops.com/case~studies.
[65] Financing through bonds may be necessary to fund and construct expensive capital improvements and qualified plant operators may need to be trained or hired.
[66] ARM 17.30.1350 contains Montana's compliance schedule authorizing provision.

Existing Use Protection and NonPoint Source Controls for the General Variances

The EPA's water quality standards regulation (40 CFR § 131.10(h)) states that:

> "States may not remove designated uses if:
> (1) They are existing uses, as defined in § 131.3, unless a use requiring more stringent criteria is added; or
> (2) Such uses will be attained by implementing effluent limits required under sections 301(b) and 306 of the Act and by implementing cost-effective and reasonable best management practices for nonpoint source control.

Existing uses are those uses actually attained in the waterbody on or after November 28, 1975, whether or not they are included in the water quality standards. Federal regulations preclude removing designated uses if they are existing uses. A variance is a time-limited designated use and criterion for a specified pollutant(s), permittee(s), and/or water body or waterbody segment(s) that reflects the highest attainable condition during the specified time period. A variance provides a mechanism to make incremental progress toward the ultimate water quality objectives for the water body.

When adopting a variance, states and authorized tribes retain the underlying designated use and criterion in their standards to apply to all other permittees not addressed in the WQS variance, to identify threatened and impaired waters under CWA Section 303(d), and to establish a Total Maximum Daily Load (TMDL). The underlying designated use and associated criteria reflect the ultimate water quality objectives for a water body. In contrast, a variance is time-limited, and reflects the highest attainable condition during a specified time period. Designated uses and existing uses represent ultimate goals independent of time, whereas the highest attainable condition during a variance represents a time-limited proximate goal with the purpose of providing a mechanism toward achieving the ultimate designated use and thus protecting the existing use. Because the underlying designated use and associated criteria remain in place for the long-term, existing uses that are protected by the underlying designated use and associated criteria are not removed when a state adopts a time-limited variance.

For the nutrient rules that the EPA is acting on today, it is clear that Montana's implementation of nutrient variances (whether general or individual) will *improve* water quality, and place many impaired Montana waters on a pathway toward full attainment. Such variances recognize the reality that nutrient loadings from existing point sources need to be reduced, and that time is needed to accomplish such reductions. Rather than removing designated uses, the EPA believes such variances are essential to achieving protection of designated uses (and attainment of base numeric criteria) by a date certain.

Unlike an action to permanently remove a designated use, Montana's general variances retain the designated use as a long-term goal. The variances are authorized for no more than 20 years, and EPA understands the state will include limits based on the NNC in the permit fact sheet. Doing so ensures that permittees remain aware of their long-term compliance goals, and demonstrates a commitment to pursue achieving the WQBELs, the underlying designated use and the NNC within a period not to exceed 20 years.

It is clear from Montana's response to public comments that the state recognizes its obligation to protect existing uses, and that variances are not authorized for new or increased dischargers if existing use(s) would be impacted. For example, consider a water body where water quality conditions for all pollutants (including nutrients) support designated uses (i.e., the designated use is an existing use). In this scenario, a new/expanded discharge that would cause or contribute to a water quality standards exceedance would

not protect an existing use and fail to comply with MCA 75-5-303(1). Thus, the EPA interprets MCA 75-5-303(1), and Montana's response to comment, as acknowledging that variances are not authorized in the circumstances described therein, and that permits for such new/expanded discharges would need to include effluent quality limitations that protect designated and existing uses *on the date such discharges are initiated.* Any such permits would also have to comply with Montana nondegradation requirements.

Regarding 40 CFR § 131.10(h)(2), Montana evaluated cost-effective and reasonable best management practices for nonpoint sources under the control of a discharger. This is consistent with §131.10(h) because Montana's general variances and individual variances provision clearly only allow variances that are discharger(s) specific versus waterbody wide. Given the scope of Montana's provisions, the EPA believes it is reasonable for the state to evaluate only those best management practices for nonpoint source control that are within the control of a discharger.[67,68,69] In the scenario where there are no nonpoint sources under the control of the discharger (which the EPA believes is often the case) then the justification for the variance need not consider what can be achieved with implementation of cost-effective and reasonable best management practices for nonpoint source control.

In developing its general variance approach, Montana considered whether land application would be a viable nonpoint source control by various dischargers. Montana also discussed water rights issues related to land application opportunities with its rulemaking workgroup in March 2010. Workgroup discussion notes document the challenges noted with land application, specifically that land application requires access to available land with reliable landowner permission; piping to transport waste to the land application area; retention zones for periods when waste cannot be land applied; and funding. Because of this host of issues, Montana determined that land application was not be a viable option for many communities as a cost-effective BMP. Land application is one of the alternatives that, per DEQ-12B, dischargers should consider as part of the facility optimization study required for all facilities. Therefore, Montana considered cost effective and reasonable BMPS for non-point sources within the control of the discharger.

It is clear from Montana's evaluation of land application options that in the typical case where waters are now impaired, implementing cost effective and reasonable BMPs for nonpoint source control alone will not attain designated uses. It is most likely that a reduction in TP and TN load from a combination of point sources and nonpoint sources will ultimately be necessary to achieve the NNC and attain designated uses in wadeable streams. Rather than removing the underlying designated use, Montana's adoption of a variance provides time, up to 20 years in this case, to attain the underlying designated use. During this interim period, Montana is committed to a process of evaluating both point source control technology and nonpoint source reductions to identify the highest attainable condition at regular intervals. The EPA fully anticipates that this process will include further examination of cost effective and reasonable BMPs for nonpoint source control. As an example, Montana has encouraged dischargers

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[67] EPA. 2011. EPA Technical Support Document for EPA's Action on the State of Oregon's New and Revised Human Health Water Quality Criteria for Toxics and Associated Implementation Provisions Submitted July 12 and 21, 2011. October 17,2011.

[68] By contrast, for variances that temporarily relax requirements for all sources in the watershed (waterbody variances), the EPA interprets the provision as requiring an assessment of all contributing nonpoint sources.

[69] 40 CFR 132, Appendix F, Procedure 2 A.3. "A WQS variance shall not be granted if standards will be attained by implementing effluent limits required under sections 301(b) and 306 of the Clean Water Act (CWA) and by the permittee implementing cost-effective and reasonable best management practices for nonpoint source control."

to evaluate nutrient trading opportunities with nonpoint source partners. Montana recently released a comprehensive report that examined the viability of nutrient trading within the state.[70]

Based on this information, the EPA is approving Montana's nutrient rules as consistent with 131.10(h).

INDIVIDUAL VARIANCES

Section 3.0

Section 3.0 of DEQ Circular 12B contains introductory information and discusses how Section 3.0 is organized. This section establishes that the final permit limit for individual variances implementing the end-of-pipe-treatment requirements and NNC will be expressed as a load. Section 3.0 is approved.

Eligibility Criteria for Individual Variances

Sections 3, 5 and 6 of ARM 17.30.660 and Section 3.1 of DEQ-12B describe the considerations for individual variances and application process. The ARM language reads as follows:

> (3) An application for an individual variance must adequately demonstrate that there are no reasonable alternatives that eliminate the need for a variance and that attainment of the base numeric nutrient standards is precluded due to economic impacts or limits of technology, or both. If the demonstration relies upon economic impacts, the department shall consider any guidance developed by the department and the nutrient work group, as provided in 75-5-313(2), MCA.
>
> (5) The department shall review each application for an individual variance to determine whether a reasonable alternative, such as trading, a permit compliance schedule, a general variance, reuse, recharge, or land application would eliminate the need for an individual variance. If the department makes a preliminary finding that a reasonable alternative to approving an individual variance is available, the department shall consult with the applicant prior to making a final decision to approve or deny the individual variance.
>
> (6) If, after consultation with the applicant, the department determines that no reasonable alternative to an individual variance exists, the department shall determine whether the information provided by the applicant pursuant to (3) adequately demonstrates that attaining the base numeric nutrient standards is not feasible. If the department finds that attaining the base numeric nutrient standards is not feasible, the department shall approve an individual variance, which will become effective and incorporated into the applicant's permit only after adoption by the department in a formal rulemaking proceeding.

Section 3.1 of DEQ-12B emphasizes many of the conditions described in ARM 17.30.660 Sections 3, 5 and 6 regarding the analysis of alternatives to a variance; basis for the individual variance; and the process for review and approval by the state. In addition, Section 3.1 provides additional details on the considerations for individual variances. For example, Section 3.1 authorizes Montana to grant individual variance limits for up to 20 years and establishes that Montana must review the economic basis for the individual variance every three years. Section 3.1 also establishes that the variance will identify the "lowest effluent concentration that is feasible based on achieving the highest attainable condition."

[70] Morrison-Maierle, Kieser and Associates; and M.J Walsh and Associates. Water Quality Trading Business Case for Montana. 2014. Report prepared for MDEQ.

Basis for Approval

The EPA's water quality standards regulation at 40 CFR § 131.13 provides that variance policies may be adopted at state discretion, and that such general policies are subject to review and approval by the EPA.[71,72] As noted in the general variance section of this letter, under the EPA's water quality standards regulation, adoption of variances may be granted if it can be demonstrated that the otherwise applicable designated use and criterion or criteria are not feasible to attain during a certain time frame. 40 CFR § 131.10(g) sets forth the limited factors that may be used to justify variances.

ARM 17.30.660(3) specifies that variances are authorized only when no reasonable alternatives to the individual variance exist. ARM 17.30.660(5) and Section 3.1 of DEQ-12B specify that the analysis should evaluate non-discharge options (e.g., pollutant reduction or elimination, seasonal retention, land application, reuse, recharge) as well as nutrient trading and the use of compliance schedules. Such a requirement to conduct a thorough evaluation of alternatives, including non-discharge options, is an important component of deciding whether the WQS is attainable or whether it is unattainable for a period of time.

In addition to requiring an analysis of alternatives to the individual variance, ARM 17.660(3) identifies three situations (eligibility criteria) where adoption of individual variances is authorized. This is in contrast to the federal rule (40 CFR 131.10(g)), which authorizes removal of designated uses in six situations. The three eligibility criteria included in Montana's nutrient rules are as follows: (1) attainment of the base numeric nutrient standards is precluded due to economic impacts; (2) attainment of the base numeric nutrient standards is precluded due to limits of technology; or (3) attainment of the base numeric nutrient standards is precluded due to both economic impacts and limits of technology.

While none of the EPA's 131.10(g) factors allows for "limits of technology" to be the sole basis for a designated use removal, such technology limits may be relevant to a demonstration provided under 40 CFR § 131.10(g) where water quality-based controls would "result in substantial and widespread economic and social impact." Section 3.1 of DEQ-12B (page 3-4) supports this approach, stating that:

> "Unlike the general variances discussed in Section 2.0, the Department will only grant an individual variance to a permittee after the permittee has made a demonstration to the Department that meeting the underlying standards would require water quality-based controls that results in substantial and widespread economic impacts."

The EPA agrees that there may be site-specific circumstances where it would be reasonable for Montana to consider adoption of discharger-specific individual variances provided the demonstration also shows that a 40 CFR § 131.10(g) factor has been met. The EPA is approving Montana's individual variance provisions explained above as a general policy under 40 CFR § 131.13. The decision to issue such an individual variance can only be made by completing a rulemaking to revise the WQS for an individual discharger applicable to a specific water body segment based on review of site-specific information. Each individual variance will be a Montana WQS rule change that must be submitted to the EPA for review and approval or disapproval pursuant to 40 CFR § 131.20(c). Accordingly, each individual

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[71] On September 4, 2013 the Agency proposed revisions to its WQS regulation that include new requirements addressing WQS variances. The comment period on the proposed rule closed on January 2, 2014.

[72] Guidance regarding State options is provided in Section 5.3 of the EPA Water Quality Standards Handbook (EPA-823-B-94-005, August 1994). http://water.epa.gov/scitech/swguidance/standards/handbook/index.cfm.

variance submitted for the EPA's review must include the Attorney General's certification and be consistent with the CWA and the EPA's implementing regulations, including all applicable public participation requirements. Thus, the EPA's review of Montana's individual variance authorizing provision need not evaluate each hypothetical variance the state may issue under ARM 17.30.660(3), (5) and (6) and consider whether such a variance would be consistent with the CWA and the EPA's implementing regulation. The EPA's approval of Montana's variance provision is not an automatic approval of any future variance the state wishes to grant pursuant to these provisions.

The EPA concludes that individual variance provisions in ARM 17.30.660(3), (5) and (6) are consistent with the EPA's requirements for individual variances. These provisions are approved.

Water Quality Protections that Apply While an Individual Variance is in Effect

Section 3.1 of DEQ-12B specifies that "the variance application will identify the lowest effluent concentration that is feasible based on achieving the highest attainable condition." In addition, ARM 17.30.660(4) and Section 3.2 of DEQ-12B address situations where reductions may be needed for one nutrient component of the NNC (e.g., TP) but not both (e.g., TP and TN). This section authorizes Montana to consider an individual variance request if the applicant can demonstrate, using water quality modeling, that designated uses are protected by focusing on a single nutrient. If the applicant can show that installing technology to address dual nutrient control would not improve water quality beyond what is projected with technology designed to reduce a single nutrient, the state will consider an individual variance for that nutrient parameter. In situations where individual variances are authorized based on this modeled demonstration, ambient monitoring is required to document designated use protection.

ARM 17.30.660(4) reads:

> "(4) The department may approve the adoption of an individual variance that specifies interim effluent limits different from those contained in general variance limits contained in Department Circular DEQ-12B (July 2014 edition), if water quality modeling demonstrates that greater emphasis on the reduction of one nutrient may achieve similar water quality and biological improvements as would the equal reduction of both nitrogen and phosphorus. The variance must provide effluent limits that reflect the lowest effluent concentration that is feasible based on achieving the highest attainable condition for the receiving water. A person shall submit the proposed effluent limits and supporting data in an application for an individual nutrient variance under (3). A person who has an individual variance with effluent limits that are based on this section shall, in each subsequent triennial review of those limits conducted pursuant to 75-5-313(7), MCA, collect and submit water quality data to demonstrate whether the biological status of the receiving water continues to justify those effluent limits."

In these situations, ARM 17.30.660(4) and Section 3.2 of DEQ-12B authorize Montana to set interim variance limits that reflect the highest attainable condition and require collection of water quality data to demonstrate that designated uses are supported. In addition to Montana's rule language, Sections 4.0 and 4.1 of Montana's implementation guidance[73] describe Montana's recommended approaches for dischargers interested in pursuing an individual variance based on water quality modeling: mechanistic modeling outputs or empirical data showing that the designated uses are being met.

[73] Page 12-13. Montana Department of Environmental Quality, 2014. Base Numeric Nutrient Standards Implementation Guidance. Version 1.0. Helena, MT: Montana Dept. of Environmental Quality.

In all scenarios, the expectation is that the interim effluent limit will reflect the lowest effluent concentration that is feasible based on the highest attainable condition.

Basis for Approval

The EPA's position is that variances must reflect the highest attainable condition for the duration of the variance.[74]

The procedures Montana has adopted for individual variances are consistent with the EPA's regulations in 40 CFR Part 131 and provides requirements that will facilitate progress towards the underlying designated use and applicable NNC. In situations where attainment of the water quality standard is not feasible for a period of time, the policy will require the highest degree of protection that *is* feasible, and that such requirements are re-examined not less than once every three years. As discussed earlier, any individual variance must be adopted through a state rulemaking and submitted to the EPA for review and approval. The EPA will base its review upon the applicable regulatory provisions at 40 CFR Part 131.

The EPA finds that ARM 17.30.660(4) and sections 3.1 and 3.2 are consistent with the CWA requirements and EPA's regulations. Any subsequent individual variance must include a demonstration consistent with the requirements in 40 CFR § 130.10, including the requirement that the state demonstrate that a 131.10(g) factor has been met. These provisions are approved.

NPDES Permits, and CWA Section 303(d) where there is an applicable variance

Generally, when a discharger is subject to a WQS variance, for the period of time when a variance is in effect, CWA National Pollutant Discharge Elimination System (NPDES) permits for discharges included in the variance will include limits (e.g., the "interim variance limits") derived from or specified by the variance. This approach is consistent with 40 CFR § 122.44(d)(1)(vii)(A) which requires WQBELs that "derive from and comply with" water quality standards. In situations where a TMDL establishes a wasteload allocation and a variance is granted, the permit should include effluent limits derived from the variance including any interim effluent limits approved in the variance. In situations where the discharger is meeting the waste load allocation defined in an approved TMDL, a variance is not needed. ARM 17.30.660(7) addresses this point.

However, regarding impairment decisions and TMDLs, CWA Section 303(d)(1)(A) requires that each State shall identify "those waters within its boundaries for which the effluent limitations required by section 301(b)(1)(A) and section 301(b)(1)(b) are not stringent enough to implement <u>any</u> water quality standard applicable to such waters" (emphasis added). Accordingly, listing decisions must consider the underlying designated use and criteria.

ALTERNATIVE VARIANCE

MCA 75-5-313(10)(a) and (b) authorize Montana to issue an "alternative" variance in situations where the discharger is an "insignificant" source of the nutrient load. Section 5.0 (page 15) of

[74] 1998 ANPRM, 78 Federal Register 54531.

Montana's Implementation Guidance explains that Montana may authorize an alternative variance if the permittee can demonstrate that meeting the general variance would not result in an environmentally significant water quality improvement. The guidance specifies that Montana will review requests for an alternative variance on a case-by-case basis.

However, Montana did not adopt any regulatory provisions related to "alternative" variances and is not part of the submission EPA received. Because EPA's approval does not include approval of such "alternative" variances, such variances are not effective for CWA purposes. As noted in the EPA's 2011 letter to Montana[75], none of the 40 CFR § 131.10(g) factors authorize variances based on *de minimus* (aka "insignificant") considerations; therefore, a variance based on a *de minimus* demonstration would not comply with the EPA's regulations. Instead, *de minimus* situations may be addressed through the development of total maximum daily load (TMDL) allocations pursuant to CWA Section 303(d). This approach is described in ARM 17.30.660(7) and addresses situations where a TMDL has been approved and the discharger meets the waste load allocation. As discussed earlier, the decision to issue such an individual variance can only be made by completing a rulemaking to revise the WQS for an individual segment based on review of site-specific information. The EPA will review any WQS variance based on the applicable requirements at 40 CFR Part 131. Absent an EPA-approved variance, the permit writer must use the NNC the EPA approved today, if applicable, to evaluate reasonable potential and, if necessary, develop limits as stringent as necessary to meet the applicable water quality standards (i.e., NNC). *See* CWA Section 301(b)(1)(C), 40 CFR § 122.44(d)

[75] Letter from Jim Martin, EPA Region 8 Regional Administrator to Richard Opper, MDEQ Director, 16 March 2011.

Appendix B – Summary of Facultative Lagoons in Montana

PUBLIC WASTEWATER SYSTEMS by TYPE (2010)

WATER POLLUTION CONTROL PROGRAM (WPCSRF)

	COMMUNITY OR DISTRICT	PLANT TYPE	DES. POP.	CURR. POP.	ACT. FLOW (MGD)	DES. FLOW (MGD)	Dis. CL UV	Permit and/or ND mode	Most Recent CPE	Most Recent INSP	Built or Upgrade
	Lagoon systems										
	Accelerated Facultative Lagoons										
1	CUT BANK	AccFL	4,432	3,171		0.643	no	MT-0020141?		Oct-02	U1996
2	GERALDINE	AccFL	361	284	0.060	0.116	no	MT-G580016	Sep-95	Oct-05	B2001
3	KEVIN	AccFL		186		0.033		MT0030244	Apr-88	Aug-94	B2003
	Facultative Lagoons										
1	ALDER (Madison CSD)	FL	260	116	0.012	0.026	UV	SI		Sep-05	B2004
2	AMSTERDAM/CHURCHILL (CSD #307)	FL	1,100	827	0.083	0.110	no	SI		Aug-05	B1977
3	ANTELOPE	FL	150	115	0.008	0.010	no	TC		Oct-98	B1986
4	AUGUSTA (Lewis&Clark CSD)	FL	460	300	0.030	0.041	no	TC		Jul-99	B2001
5	BAINVILLE	FL	283	153	0.020	0.035	no	SI		Jul-04	B2009
6	BAKER	FL	3,200	1,793	0.170	0.350	no	MT-G580029 w/SI		May-03	U1989
7	BEARCREEK	FL		68		0.008	no	TC		Nov-03	
8	BELFRY	FL		250		0.030	no	TC		Jun-00	B1969/U2000
9	BELLE CREEK	FL		100		0.070					
	BOULDER	FL	2,500	1,665	0.105	0.250	no	MT-0023078		Sep-94	U1964
10											
11	BOX ELDER (Hill CSD)	FL		108		0.019	no	TC		Apr-00	B1998
12	BRADY (PONDERA CO. COMM.)	FL	559	170	0.031	0.062	no	MT-G580022		Aug-95	B1955/U60s
13	BROADUS	FL	1,000	451	0.045	0.100	no	MT-G580015		Sep-04	B1967/U1980
14	BROADVIEW	FL	200	150	0.015	0.020	no	IP		Jul-03	B1969
15	BROCKTON (EPA TRIBAL)	FL		491		0.037	no	MT-0023001			U1987
16	BUSBY (EPA/TRIBAL)	FL		100		0.010		ND?			
17	CARTER (Choteau CWD)	FL		75		0.010	no	TC		Dec-92	
18	CASCADE	FL		930		0.097	no	SI		Nov-07	New 1998
19	CHARLO (LAKE CW&SD)	FL/M	815	452	0.047	0.086	UV	EPA and CW		Aug-07	B2005
20	CHERRY CREEK WUA (Valley Co. SID #2)	FL		175				TC		Apr-92	B1965
21	CHESTER	FL	1,100	871	0.067	0.168	no	MT-0020338	Jun-99	Aug-03	U1984
22	CHOTEAU	FL	2,400	1,749		0.300	no	MT-0020052 w/SI		May-94	B1954
23	CIRCLE	FL	2,000	644	0.065	0.160	no	MT-0020796		Nov-02	U1974
24	CULBERTSON	FL		716	0.059		no	IP		Jul-04	B1977
25	CUSTER (YELLOWSTONE CSD)	FL	480	145	0.048	0.015	no	TC		May-03	B1968
26	DARBY	FL	707	679	0.135	0.150	no	MT-G580011		Oct-02	U1980
27	DEERFIELD COLONY (Lewistown)	FL		100		0.010	no	SI			
28	DENTON	FL	440	293	0.030	0.044	no	MT-0022462		Nov-04	B1999
29	DIXON (EPA/TRIBAL)	FL		100		0.010		TC?			
30	DODSON	FL	142	107	0.230	0.030	no	TC		Apr-00	B1958/U2007
31	DRUMMOND	FL		282		0.008	no	MT-G580002		Aug-93	U2002
32	DUTTON	FL	500	401	0.030	0.050	no	MT-G580023	Jul-01	Apr-00	**B1973**
33	EDGAR (Carbon CSD)	FL	200	100	0.020	0.010	no	MT-G000043		Nov-03	B1967
34	FAIRFIELD	FL		715		0.100		MT-G580003		Sep-94	B1958
35	FAIRVIEW	FL	1,693	709	**0.194**	0.160	no	NDG125356		Jun-05	U1985
36	FALLON (PRAIRIE CSD)	FL	300	138	0.014	0.030	no	MT-G580025 w/TC		May-03	B1969
37	FLAXVILLE	FL		83		0.017	no	IP		May-00	
38	FOREST PARK MHP (near Whitefish?)	FL									
39	FORT PECK HOUSING AREA	FL	545	281	0.051	0.098	no	MT-G580019	Jul-94	Sep-05	U1995
40	FRAZER (EPA/TRIBAL)	FL		100		0.040		TC?			
41	FROID	FL		230		0.040	no	MT-G580028	Jul-96	Jul-94	B1995
42	GEYSER	FL	192	144	0.014	0.019	no	SI		Feb-02	B1983
43	GILDFORD (Hill CSD)	FL	315	150	0.015	0.022	no	TC		Aug-03	B1981
44	GRASS RANGE	FL		166		0.035	no	MT-0030309		Mar-96	
45	HARRISON (Madison CW&SD)	FL	180	162	0.017	0.024	no	SI		Sep-05	B2000
46	HAYS (EPA/TRIBAL)	FL		100				TC?			
47	HINGHAM	FL	400	180	0.018	0.040	no	TC		Aug-03	B1954/U1963
48	HOBSON	FL	285	240	0.024	0.045	no	MT-0021636	Sep-95	Mar-04	B1994
49	HYSHAM	FL		410		0.100	no	MT-0021709	Aug-90		
	INVERNESS	FL		100		0.020	no	TC			
50											
51	JOPLIN (LIBERTY CSD)	FL		100			no	SI			
52	JORDAN	FL		550		0.800	no	**Permit#?**		Nov-02	
53	JUDITH GAP	FL	240	164	0.036	0.046	no	SI		Aug-04	B2001
54	LAMBERT	FL	**150**	**150**	**0.015**	**0.015**	no	**TC**		Sep-05	B1971
55	LAVINA	FL	181	209		0.022	no	MT-G580013	Jul-97		B2002
56	LINCOLN (LEWIS & CLARK CSD)	FL		1,113		0.125		SI		Oct-99	U1999
57	LODGE GRASS (EPA/TRIBAL)	FL		504		0.080		EPA			
	MEDICINE LAKE	FL		350		0.033		MT-G580024			
58											
59	MELROSE	FL		130		0.013	no	SI	Oct-94	May-00	
60	MELSTONE	FL	400	136	0.018	0.040	no	TC		Oct-02	B1974
61	MOORE	FL	277	184	0.028	0.018	no	SI		Mar-04	B1983
62	NASHUA	FL	435	325	0.042	0.048	no	MT-G580034		Oct-04	B2003
63	OPHEIM	FL		144		0.060	no	TC		May-00	B1955
64	OUTLOOK (SHERIDAN CSD)	FL		100			no	MT-G580026		Apr-97	
	PEARSON ADDITION (Cascade Co.) and	FL		65	0.006		no	TC		Jun-93	
65	TRAILER TERRACE MHP										
66	PHILIPSBURG	FL		1,000		0.200		MT-G580005	Jun-95	Jun-93	
67	PLENTYWOOD	FL	4,000	2,047	0.270	0.420	no	MT-G580008		Oct-05	B1973
68	PLEVNA	FL	400	200	0.040	0.020	no	TC		May-03	
69	POWER (TETON CW&SD)	FL		100		0.020	no	TC		Nov-03	
70	RAMSAY	FL		125		0.020	no	IP or TC?		Oct-96	

PUBLIC WASTEWATER SYSTEMS by TYPE (2010)

WATER POLLUTION CONTROL PROGRAM (WPCSRF)

	COMMUNITY OR DISTRICT		PLANT TYPE	DES. POP.	CURR. POP.	ACT. FLOW (MGD)	DES. FLOW (MGD)	Dis. CL UV	Permit and/or ND mode	Most Recent CPE	Most Recent INSP	Built or Upgrade
71	REED POINT	FL			150		0.017	UV	SI		Mar-04	B1994
72	RESERVE	FL			75		0.005	no	TC		Jun-93	
73	RICHEY	FL		560	250	0.450	0.110	no	SI		Jun-96	B2006
74	ROBERTS (CARBON CSD)	FL		340	300	0.110	0.230	no	SI (alfalfa)		Nov-03	B1969
75	ROSEBUD (Rosebud CSD)	FL		500	100	0.050	0.010	no	MT-0022373 TC?		May-03	B1978
76	ROY (Fergus CW&SD)	FL			125				SI		Jun-97	
77	RUDYARD (Hill CSD)	FL		600	275	0.028	0.060	no	SI		Aug-03	B1954
78	RYEGATE	FL		500	268	0.027	0.050	no	MT-0020451		May-04	B1969
79	SACO	FL			180		0.033	no	MT-G580012		Apr-00	
80	SAVAGE (RICHLAND CSD)	FL			300		0.040		MT-0024783		Jun-96	
81	SCOBEY	FL			1,154		0.420		TC		Apr-97	B1968
82	SHELBY	FL			3,419		0.330		MT-G580006		Aug-03	
83	SHERIDAN	FL		600	**652**	**0.099**	0.060	no	MT-0022098		Sep-03	B1960
84	SIDNEY	FL			5,286		0.800		IP		Oct-94	
85	SIMMS	FL		370	200	0.020	0.037	no	SI		Sep-04	B1980
86	STANFORD	FL		586	454	0.057	0.074	no	MT-0022161	Apr-93	Jul-06	B2003
87	STOCKETT	FL		240	220	**0.035**	0.034	UV	MT-0030091	Aug-99	Apr-07	B1994
88	SUNBURST	FL			450		0.514		MT-0021679	Apr-98	Jun-98	B1984
89	SWEET GRASS (TOOLE CSD)	FL		150	125	0.015	0.020	UV	MT-G580010		Jun-96	B2004
90	TERRY	FL			627		0.150		MT-G580017		Jul-99	
91	THREE FORKS	FL		2,200	1,728	0.406	0.453	no	MT-0020401 & IP		Sep-98	B1982
92	TRAILER TERRACE MHP Cascade Co	FL						no	TC		Jun-93	
93	TURNER	FL		90	90	0.011	0.011	no	IP	Aug-99	Mar-07	B1992
94	TWIN BRIDGES	FL		600	407	**0.080**	**0.070**	no	MT-0028797	Jan-91	Sep-05	U1990
95	VIRGINIA CITY/NEVADA CITY	FL			150		0.015		SI		Oct-96	B2004
96	WARM SPRINGS (MT. ST. HOSPITAL)	FL			1,351		1.108		MT-G580004		Jun-96	
97	WESTBY	FL		350	151	0.018	0.025	no	TC		Aug-05	B1973
98	WHITE SULPHUR SPRINGS	FL		1,800	984	**0.150**	**0.180**	no	MT-G580021		Nov-05	B1958
99	WHITEHALL	FL		1,480	1,184	0.100	0.250	UV	MT-0020133		Apr-00	B1960
100	WHITEWATER (Phillips CSD)	FL			100		0.010		TC			
101	WILLOW CREEK (Gallatin CSD)	FL			100		0.015	Cl	MT-0025038	Jun-85	Mar-89	
102	WINIFRED	FL			200		0.020		TC		Oct-05	
103	WISDOM	FL			135		0.015		TC or SI	Aug-95	Jun-95	
104	WOODCOCK (Flathead IR)	FL			140		0.014		No Permit	Jul-95		
105	WORDEN-BALLANTINE W&S DIST.	FL			680		0.114		GW XXXXX or TC		Aug-94	
	TOTAL				51,931		11.181					

Appendix C – Facultative Lagoon Technology Summary Table

FACULTATIVE LAGOON TECHNOLOGY MATRIX

Treatment	Water Quality Function	Ancillary Benefits	Risks/Other Considerations	Dependencies	Complimentary treatments	Capital Cost	Maintenance Requirements	Footprint	Examples in Montana	Effective in low temperatures	Patent
EXAMPLE - TECHNOLOGY A	*• Reduces N* *• Reduces BOD* *• Functions as a catalyst for X*	*• Ancillary benefits*	*• Potential risks associated with the example treatment that may increase management levels, be a health concern, etc.*	*• Other treatments the example treatment requires to function*	*• Treatments that would work well in conjunction with the example treatment*	*Low ($), Medium ($$), High ($$$), Very High ($$$$)*	*None/Low/Med/High*	*No Increase/ Moderate increase/ Large increase*	*Yes/ No*	*Yes/No*	*Yes/No*
OPERATIONAL											
Retention Times Fill/Draw	• Improves removal efficiency of P, N, BOD, TSS, metals, coliform bacteria, and pathogens	• Increases retention time • Reduces short-circuiting • Facilitates optimal treatment depths	• Effluent water quality must be monitored.	None	All operational BMPs can be combined with other lagoon optimization technologies	$	Low	No Increase	Yes	Yes	No
Solid Waste Removal	• Improves removal efficiency of P, N, BOD, TSS, metals, coliform bacteria, and pathogens	• Increases capacity • Minimizes wet weather problems and overflows	• Damage to lagoon liner • Scouring of lagoon embankments	None	All operational BMPs can be combined with other lagoon optimization technologies	$$	Low	No Increase	Yes	Yes	Yes - Numerous existing and patented technologies
Reduce Influent	• Minimizes sewer overflows • Improves removal efficiency of P, N, BOD, TSS, metals, coliform bacteria, and pathogens	• Saves capacity • Eliminates wet weather problems and overflows	• Cost/benefit of reducing influent should be carefully evaluated.	None	All operational BMPs can be combined with other lagoon optimization technologies	$$ to $$$$	Low	N/A	Yes	Yes	Yes - some sliplining technologies are patented
Mosquito Control	• No significant treatment of primary pollutants	• Public health benefits	• Chemicals used can be harmful to other insects and aquatic habitat if not applied properly, and repeated use could cause resistance.	None	Mowing of vegetation along embankments, vegetation harvesting	$ to $$	Low	N/A	Yes	No	Yes - Some treatment technologies (Larvasonic)
Maintenance Schedule/Activities	• Improves removal efficiency of P, N, BOD, TSS, metals, coliform bacteria, and pathogens	• Ancillary benefits are dependent on specific maintenance activity	None	• Identification of proper maintenance activities and schedules may require monitoring and technical evaluation. •Activities require varying levels of operator time and skill to be effective.	All operational BMPs can be combined with other lagoon optimization technologies	$	Low-High	No Increase	Yes	Yes	No
MECHANICAL											
Aeration	• Removal of Ammonia-N, nitrite, TSS, & BOD. • Improves TSS breakdown by mixing.	• Odor reduction if applied appropriately.	Risk of oxygenating anaerobic zone.	None	Aeration can be used with most lagoon optimization technologies.	$$$	Low-Med	No Increase	Yes	Yes	Yes, Numerous existing and patented technologies.
Lagoon Covers	• Reduces TSS • Provides insulation to maintain higher temperatures, thereby increasing lagoon efficiency	• No impediment to aeration equipment pond; • Prevent UV light penetration & surface oxygenation, maintaining anaerobic conditions; • Quick and simple to install; • Spread automatically as the liquid levels rise and fall; • Deters birds from landing on the lagoon and contributing N • Virtually indestructible UV resistant HDPE material; • Cannot tear like conventional membrane covers; and • Can freeze into ice covered lagoons without damage.	None	None	Lagoon covers can be used with most lagoon optimization technologies.	$ to $$$	Low	No Increase	Yes	Yes	Yes
Floating Barriers	• Reduces TSS • Improves BOD and prevents lagoon short-circuiting • Compatible with other technologies.	• No impediment to aeration equipment pond; • Quick and simple to install; • Virtually indestructible UV resistant HDPE material; • Can freeze into ice covered lagoons without damage; • Automatically adjusts to changes in water volume;	None	None	Floating Barriers can be used with most lagoon optimization technologies.	$ to $$$	Low	No Increase	Yes	Yes	Yes
Fixed Film Bioreactors • Fixed Media Systems o Trickling filter o Rotating Biological Contactor (RCB) o Packed Bed (Fixed Bed) Reactors o Submerged Aerated Reactors • Suspended (Free Floating) Media Systems o Moving Bed Bio-Reactors (MBRR) o Fluidized Bed Reactors	• Reduces Ammonia • Reduces Total Nitrogen • Reduces BOD • Reduces TSS • Reduces Total Phosphorous • Nitrification • Denitrification	• It is stable under load variations, insensitive to temporary limitation and provides consistent treatment results. • It generates low solids and requires no or minimum polymer for solid/liquid separation. • It requires a small footprint compared to other processes. Typically it requires 1/3rd the space required for Activated Sludge Process. • It requires a low capital cost and is comparable to cost of Activated Sludge Process. It is cheaper than the MBR process. • Existing wastewater plants can be upgraded easily with many fixed film technologies. New fixed film plants can be upgraded to handle higher loads with no or minimum cost and construction. • It is easy to operate, has automatic sludge wasting, has no sludge Return and no MLSS, and there no issue of media clogging. • Capacity can be easily upgraded by simply increasing the fill fraction of biofilm carriers. • No return activated sludge stream required. • Actively sloughed biofilm automatically responds to load fluctuations.	• Groundwater and/or drinking water contamination.	Varies depending on the reactor configuration and/or the required treatment.	Fixed Film Bioreactors can easily be incorporated with most other identified lagoon optimization technologies.	$$$ to $$$$	Low-High, depending on the system and the treatment needs.	No Increase to Potential increase to existing pond modification.	Yes	Yes	Yes, Numerous existing and patented technologies.

Activated Sludge	• Reduces Ammonia • Reduces Total Nitrogen • Reduces BOD • Reduces TSS • Reduces Total Phosphorous • Nitrification • Denitrification	• It is stable under load variations, insensitive to temporary limitation and provides consistent treatment results. • Existing wastewater plants can be upgraded easily with many fixed film technologies. New fixed film plants can be upgraded to handle higher loads with no or minimum cost and construction. • Actively sloughed biofilm automatically responds to load fluctuations.	• Groundwater and/or drinking water contamination.	The Activated Sludge process is a complete system that requires many components to work effectively.	The Activated Sludge process is a complete system that requires many components to work effectively.	$$$$$	Very High	Potential increase or redesign of existing footprint	Yes	Yes	Yes, Numerous existing and patented technologies.
Phosphorus Removal • chemical precipitation; • enhanced biological uptake; • tertiary treatment; and • Activated Alumina Adsorption.	• Permanent removal of P.	• Provides diverse aquatic and terrestrial habitat.	• Groundwater and/or drinking water contamination.	• Previous water quality treatment.	Can be an add-on technology to numerous identified technologies.	$$ to $$$	High	No Increase	Yes	Yes	Yes, Numerous existing and patented technologies.
Alternate Carbon Sources	• Removal of TN (Denitrification)		• Groundwater and/or drinking water contamination.	• Must have nitrified wastewater in an anaerobic environment.	• Aeration, Activated Sludge, Fixed Film	$$ to $$$	Med-High	No Increase	Yes	Yes	Yes, Numerous existing and patented technologies.
Pathogens/Disinfection	• Permanent removal harmful pathogens.		• May harm beneficial microorganism colonies.	• Previous water quality treatment.	Can be an add on technology to numerous identified technologies.	$$	Med	No Increase	Yes	Yes	Yes, Numerous existing and patented technologies.
ECOLOGICAL											
Constructed Wetlands	• Vertical wetlands primarily address NH_4^+. • Subsurface wetlands primarily address NH_4^+ and NO_3^-. • Surface wetlands primarily address NO_3^-. • All wetlands are effective at addressing BOD and TSS. • P is stored in plant tissue and sediments.	• Provides diverse aquatic and terrestrial habitat.	• N may increase as a result of water fowl use. • Mosquitoes. • Odor. • Invasive plant species.	• Previous water quality treatment.	N/A	$$	Low	Large	Yes	No	No
Floating Island	• Effective at NH_4^+ removal under aerated conditions. • Effective at NO_3^- removal without aeration. • P is stored in plant tissue and sediments. • Reduction of TSS.	• Provides diverse aquatic and terrestrial habitat.	• Sulphide toxicity. • Increased ice formation. • Potential to attract waterfowl. • Invasive plant species.	• Previous water quality treatment.	• Aeration	$$ to $$$	Low	No Increase	Yes	No	Yes
Vegetation Harvesting	• Permanent removal of P, • Removal of N, BOD, TSS, metals, coliform bacteria, and pathogens	• Can be used as livestock feed, poultry feed, compost or biofuel.	• Toxic types of blue-green algae may grow with duckweed. • Some harvesting methods can contain heavy metals, limiting reuse options.	• Previous water quality treatment.	N/A	$	High	No Increase	No	No	No
Terrestrial Application	• Removal of P, N, BOD, TSS, metals, coliform bacteria, and pathogens	• Can provide lumber or forage or other economic resources.	• Groundwater and/or drinking water contamination.	• Previous water quality treatment.	N/A	$	Med	Large	Yes	Yes	No
Vermiculture	• Removal of N, BOD, metals, coliform bacteria, chemicals and pathogens	• Can provide compost.	• By products of vermicomposting may contain toxins and pathogens. • High temperatures will kill earthworms.	N/A	N/A	$$ to $$$	Low	Moderate	No	Yes	Yes
Fish	• Permanent removal of P. • Removal of N and BOD	• Harvested fish can be used as pet food or compost.	• Die off may occur if pollutant levels are too high.	• Previous water quality treatment.	N/A	$	Med	No Increase	No	Yes	No
Eco-Machines™	• Capable of processing all pollutants depending on the included components.	• Can be an educational resource and/or tool.	• Depends on included components.	N/A	N/A	$$$	Med	Moderate to Large	No	Yes	Yes