November 7, 2018

Representatives
Elm Creek Watershed Management Commission
Hennepin County, MN

The meeting packets for these meetings may be found on the Commission’s website: http://www.elmcreekwatershed.org/minutes–meeting-packets.html

Dear Representatives:

A regular meeting of the Elm Creek Watershed Management Commission will be held on Wednesday, November 14, 2018, at 11:30 a.m. in the Mayor’s Conference Room at Maple Grove City Hall, 12800 Arbor Lakes Parkway, Maple Grove, MN.

The Technical Advisory Committee (TAC) will meet at 10:00 a.m., prior to the regular meeting to discuss two items:

1) Identify, prioritize, and undertake subwatershed assessments with possible assistance from the Commission
2) Use of wetlands for irrigation purposes.

TAC meeting materials can also be found on the Commission’s website.

Please email Tiffany at tiffany@jass.biz to confirm whether you or your Alternate will be attending the TAC and the regular meetings.

Thank you.

Regards,

Judie A. Anderson
Administrator
JAA:tim

Encls:  Meeting Packet
cc:   Alternates   HCEE   Jeff Weiss   BWSR
      TAC Members   TRPD   Diane Spector   DNR
      City Clerks   MPCA   Met Council   Official Newspaper

2:\Elm Creek\Meetings\Meetings 2018\11 Notice_reg and TAC meetings v2.docx
AGENDA
Technical Advisory Committee and Regular Meetings
November 14, 2018

1. Call TAC meeting to Order.
   a. Approve agenda.*
   b. Approve Minutes of last TAC meeting.*

2. Identify, prioritize, undertake SWAs with possible assistance from the Commission
   b. SWA Cost Share Application.*
   c. Current Cost Share Policy.*

3. Use of wetlands for irrigation purposes.
   a. Using Wetlands as Irrigation Ponds.*
   b. Iron in Wetland Systems.*
      1) Treatment Wetlands.*
      2) Wetlands Mitsch and Gosselink.*

4. Ongoing Compliance Requirements for Buffer Law. – Barta.

5. Other Business.

6. Adjourn meeting of TAC.

---

1. Call Regular Meeting to Order.
   a. Approve Agenda.*

2. Consent Agenda.
   a. Minutes last Meeting.*
   b. Treasurer’s Report and Claims.**

3. Open Forum.
   a. Presentation – Plymouth Reach D.

4. Action Items.
   a. Project Reviews – see Status Report.*
   b. Closed Project Account Policy.*

---

* in meeting packet
** available at meeting
c. Fish Lake Alum Treatment RFP. **
d. Fish Lake SWA – Cooperative Agreement.*
e. Opportunity Grant Application – Fish Lake Alum Treatment.*
f. Local Plans.
   1) Maple Grove.*
   2) Plymouth.*
   3) Corcoran.
g. BWSR Watershed-Based Funding Grant Agreement.*
h. Hennepin County GIS User Agreement.*

5. Old Business.


7. Communications.
a. Hennepin County Approves Levies.*

8. Education.
a. WMWA Update.**

9. Grant Opportunities and Updates.
a. FEMA Floodplain Mapping – see Staff Report.
b. Diamond Lake SWA Grant Application – see Staff Report.
c. North Fork Rush Creek SWA Implementation – see Staff Report.

10. Project Reviews – also see Staff Report.*

<table>
<thead>
<tr>
<th>10. Project Reviews. (See Staff Report.*)</th>
</tr>
</thead>
<tbody>
<tr>
<td>b.</td>
</tr>
<tr>
<td>c.</td>
</tr>
<tr>
<td>d.</td>
</tr>
<tr>
<td>e.</td>
</tr>
<tr>
<td>f.</td>
</tr>
<tr>
<td>g.</td>
</tr>
<tr>
<td>h.</td>
</tr>
<tr>
<td>i.</td>
</tr>
<tr>
<td>j.</td>
</tr>
<tr>
<td>k.</td>
</tr>
<tr>
<td>l.</td>
</tr>
<tr>
<td>m.</td>
</tr>
<tr>
<td>n.</td>
</tr>
<tr>
<td>o.</td>
</tr>
<tr>
<td>p.</td>
</tr>
<tr>
<td>q.</td>
</tr>
<tr>
<td>r.</td>
</tr>
<tr>
<td>s.</td>
</tr>
</tbody>
</table>

* in meeting packet
** available at meeting
<table>
<thead>
<tr>
<th>Item</th>
<th>Year</th>
<th>Code</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>t.</td>
<td>2017-050W</td>
<td>Ernie Mayer Wetland/floodplain violation, Corcoran.</td>
<td></td>
</tr>
<tr>
<td>u.</td>
<td>2018-001</td>
<td>Rush Creek Commons, Maple Grove.</td>
<td></td>
</tr>
<tr>
<td>v.</td>
<td>2018-004</td>
<td>Rush Creek Restoration, Maple Grove.</td>
<td></td>
</tr>
<tr>
<td>w.</td>
<td>2018-005</td>
<td>Sundance Greens, Dayton.</td>
<td></td>
</tr>
<tr>
<td>x.</td>
<td>2018-014</td>
<td>Refuge at Rush Creek, Corcoran.</td>
<td></td>
</tr>
<tr>
<td>y.</td>
<td>2018-018</td>
<td>Summers Edge Phase II, Plymouth.</td>
<td></td>
</tr>
<tr>
<td>z.</td>
<td>2018-020</td>
<td>North 101 Storage, Rogers.</td>
<td></td>
</tr>
<tr>
<td>aa.</td>
<td>2018-021</td>
<td>113th Lane Extension/Brockton/101, Rogers.</td>
<td></td>
</tr>
<tr>
<td>ab.</td>
<td>2018-026</td>
<td>Windrose, Maple Grove.</td>
<td></td>
</tr>
<tr>
<td>ac.</td>
<td>2018-027</td>
<td>CR202 Bridge, Dayton.</td>
<td></td>
</tr>
<tr>
<td>ad.</td>
<td>2018-028</td>
<td>Tricare Third Addition, Maple Grove.</td>
<td></td>
</tr>
<tr>
<td>af.</td>
<td>2018-032</td>
<td>Encore, Corcoran.</td>
<td></td>
</tr>
<tr>
<td>ag.</td>
<td>2018-033</td>
<td>Cloquet Island Estates, Dayton.</td>
<td></td>
</tr>
<tr>
<td>ah.</td>
<td>2018-035</td>
<td>Edgewater East, Maple Grove.</td>
<td></td>
</tr>
<tr>
<td>ai.</td>
<td>2018-037</td>
<td>Elm Creek Stream Restoration Reach D, Plymouth.</td>
<td></td>
</tr>
<tr>
<td>aj.</td>
<td>2018-038</td>
<td>Vincent Woods of Roger.</td>
<td></td>
</tr>
<tr>
<td>ak.</td>
<td>2018-040</td>
<td>Rush Creek Commons Phase II, Maple Grove.</td>
<td></td>
</tr>
<tr>
<td>al.</td>
<td>2018-043</td>
<td>BeeHive Homes, Maple Grove.</td>
<td></td>
</tr>
<tr>
<td>am</td>
<td>2018-044</td>
<td>OSI Phase II, Medina.</td>
<td></td>
</tr>
<tr>
<td>ao.</td>
<td>2018-046</td>
<td>Graco, Rogers</td>
<td></td>
</tr>
<tr>
<td>ap.</td>
<td>2018-047W</td>
<td>Old Settlers Road Wetland Delineation, Corcoran.</td>
<td></td>
</tr>
<tr>
<td>aq.</td>
<td>2018-048</td>
<td>Faithbrook Church Phase 2, Dayton.</td>
<td></td>
</tr>
<tr>
<td>as.</td>
<td>2018-050</td>
<td>Janice Little Bridge Replacement, Corcoran.</td>
<td></td>
</tr>
<tr>
<td>at.</td>
<td>2018-050</td>
<td>C&amp;D Order 9120 Train Haven Road, Corcoran.</td>
<td></td>
</tr>
<tr>
<td>au.</td>
<td>2018-050</td>
<td>C&amp;D Order 9120 Train Haven Road, Corcoran.</td>
<td></td>
</tr>
<tr>
<td>av.</td>
<td>2018-050</td>
<td>C&amp;D Order 9120 Train Haven Road, Corcoran.</td>
<td></td>
</tr>
</tbody>
</table>

A = Action item  E = Enclosure provided  I = Informational update will be provided at meeting  RPFI - removed pending further information  R = Will be removed  RP= Information will be provided in revised meeting packet..... D = Project is denied  AR awaiting recordation  

11. Other Business.

I. A meeting of the Technical Advisory Committee (TAC) for the Elm Creek Watershed Management Commission was convened at 10:03 a.m., Wednesday, April 11, 2018 in the Mayor’s Conference Room, Maple Grove City Hall, 12800 Arbor Lakes Parkway, Maple Grove, MN.

In attendance were: Todd Tuominen, Champlin; Kevin Mattson, Corcoran; Sarah Nalven, Wenck Associates, Dayton; Rick Lestina and Mark Lahtinen, Maple Grove; Kaci Fisher, Hakanson-Anderson, Medina; Ben Scharenbroich, Plymouth; Andrew Simmons, Rogers; James Kujawa and Kirsten Barta, Hennepin County Dept. of Environment and Energy (HCEE); Brian Vlach, Three Rivers Park District (TRPD); Jeff Weiss, Barr Engineering; and Judie Anderson and Amy Juntunen, JASS.

Also present: Sharon Meister, Corcoran, and Doug Baines, Dayton.

II. Motion by Lestina, second by Scharenbroich to approve the agenda. Motion carried unanimously.

Motion by Scharenbroich, second by Lestina to approve the minutes of the February 14, 2018 TAC meeting. Motion carried unanimously.

III. Commission Cost Share Policy.

At their December 13, 2017 meeting, the Commissioners discussed the “cap” on the maximum annual levy for Capital Improvement Projects (CIPs). According to the Commission’s current Cost Share Policy which was adopted in 2012, the cap is $250,000/project, $500,000/year. In December 2017 the CIP showed estimated costs for projects anticipated to be levied in 2018/payable 2019 of $1,395,250. Commissioners and TAC members were encouraged to discuss possibly raising the annual cap with their city personnel/councilors.

After discussion it was a consensus of the members of the TAC to maintain the cap at the current $250,000/project, $500,000/year. This recommendation will be brought to the Commission.

IV. Capital Improvement Program.

A. The 2018 CIP spreadsheet (Table 4.5_2017 with 2018 submittals_Rev3) has been revised as follows:

1. Stone’s Throw Wetland (line 21) has been moved from 2018 to 2019.
2. Ranchview Wetland Restoration, Maple Grove (line 23) has been moved from 2018 to 2019.
3. Hickory Drive Stormwater Improvement, Medina, (line 37) has been added in 2019.
4. Southeast Corcoran Wetland Restoration, Corcoran, (line 38) has been added in 2019.
5. Downtown Regional Stormwater Improvement, Corcoran, (line 39) has been added in 2019.
6. Elm Creek Stream Restoration Phase III, Champlin, (line 40) has been added in 2018.
7. Downs Road Trail Raingarden, Champlin, (line 41) has been added in 2018/2019.
8. Elm Creek Stream Restoration Phase IV, Champlin, (line 42) has been added in 2019
9. Lowell Pond Raingarden, Champlin, (line 43) has been added in 2019.

B. At the February 14, 2018 TAC meeting projects were reviewed for timeliness and some construction dates adjusted. Generic projects were extended out to the 2020-2024 timeframe. As a result $500,000 in projects were recommended for levy funding in 2018/payable 2019 as follows:

1. Rush Creek Main Stem Stream Restoration (line 16), Maple Grove, $75,000
2. Elm Creek Stream Restoration Reach D (line 18), Plymouth, $212,500
3. Mill Pond Gardens (line 30), Champlin, $100,000
4. Elm Creek Stream Restoration Phase III (line 40), Champlin, $100,000
5. Downs Road Trail Rain Garden (line 41), Champlin, $12,500

C. An additional CIP application has been received for Rush Creek Headwaters SWA BMP Implementation. Project cost is $200,000, with the Commission’s share of $50,000, to be constructed in 2020. Motion by Scharenbroich, second by Nalven to add this CIP as proposed. Motion carried unanimously.

Motion by Kujava, second by Scharenbroich to recommend to the Commission a call for a public meeting on May 9, 2018 to adopt a Minor Plan Amendment to incorporate these revisions/additions to the CIP. Motion carried unanimously.

[Tuominen arrived 10:10 a.m.]

D. Feasibility Reports.

1. Rush Creek Main Stem Stream Restoration,* Maple Grove, $75,000. Motion by Scharenbroich, second by Fisher to recommend moving this project forward for funding under the 2018 levy. Motion carried unanimously. John Smythe will provide the missing nutrient reduction figures.

2. Elm Creek Stream Restoration Reach D,* Plymouth, $212,500. Motion by Fisher, second by Nalven to recommend moving this project forward for funding under the 2018 levy. Motion carried unanimously.

3. Mill Pond Gardens,* Champlin, $100,000. Upon the request of the City of Champlin, motion by Scharenbroich, second by Lestina to move this project to 2019. Motion carried unanimously.

4. Elm Creek Stream Restoration Phase III,* Champlin, $100,000. Motion by Scharenbroich, second by Fisher to recommend moving this project forward for funding under the 2018 levy. Motion carried unanimously. WSB will provide the missing nutrient reduction figures.

[Barta arrived 10:28 a.m.]

5. Downs Road Trail Rain Garden,* Champlin, $12,500. Motion by Scharenbroich, second by Lestina to recommend moving this project forward for funding under the 2018 levy. As a result of the moving of the Mill Pond Gardens project to 2019, full 2018-2019 funding ($75,000) is recommended in 2018. Motion carried unanimously. WSB will provide the missing nutrient reduction figures.

[Simmons arrived 10:37 a.m.]

Motion by Scharenbroich, second by Fisher to recommend to the Commission ad valorem funding of projects 1, 2, 4, and 5, and a call for a public hearing to take public comment. Motion carried unanimously. [NOTE, IN ALL CASES THE INTENT IS TO REQUEST LEVY FUNDS IN 2018, WITH RECEIPT OF REQUESTED FUNDS IN 2019.]

E. BWSR Pilot Funding Project.

A third “pre-meeting” of watershed administrators was held March 30, 2018 to discuss options for allocating the Metro Area One Water One Plan (1W1P) Watershed-Based Funding Pilot Program dollars over the next few years. In addition, a Mississippi Basin committee and a chloride committee met to discuss potential programming. Juntunen attended for the watersheds. As a reminder, Hennepin County will be receiving $1.018 million for the next two years, to be expended by December 31, 2021. The group will refine options and develop recommendations for the formal planning meeting, 1:30 p.m., May 16, 2018, Plymouth Library, to which all eligible entities will be invited by Hennepin County.

The Mississippi Basin watersheds will be meeting once more before the formal planning meeting to refine the project prioritization criteria. Each WMO submitted up to two priority projects which will be ranked by those criteria. The chloride committee will also be bracketing the potential chloride management costs so the recommended “amount off the top” is known going into the planning meeting. They will also be checking in with other counties to see if there is interest in pursuing a multi-county or Metro Area approach.
Juntunen and Kujawa worked together to determine which projects should be submitted for this first round of watershed-based funding. They selected the Ranchview Wetland Restoration Project and the Hickory Drive Stormwater Improvement (lines 23 and 37). Motion by Lestina, second by Scharenbroich to approve these selections.  *Motion carried unanimously.*

Scharenbroich noted that an emphasis should be placed on chloride-based projects/activities. Juntunen responded that during the first year 5% funding is anticipated, with that number increasing to 10% in the second year as activities are identified.

V. Draft Manure Management Model Ordinance/Policy.

Barta presented a draft Livestock Management Policy.* It is based on the one adopted by the Pioneer-Sarah Creek WMO. That policy references the City of Greenfield’s Ordinance 2016-02* that pertains to livestock and domestic farm animals and the City of Medina’s Manure Management Policy* and related ordinances. TAC members are requested to review these documents with their cities and to come back to the next TAC meeting with comments.

VI. Aquatic Vegetation Management.

Included in the meeting packet is the final draft of the Shingle Creek Watershed Management Commission’s Submerged Aquatic Vegetation (SAV) Management Policy.* At an earlier meeting, Commissioners requested that Staff contact other WMOs with which they work to present the draft policy as a preliminary draft for consideration. Anderson reported that the Pioneer-Sarah Creek WMO was disinclined to consider an SAV policy at this time. This item will be postponed to a future TAC meeting.

VII. There being no further business, the meeting of the Technical Advisory Committee was adjourned at 11:30 a.m. The TAC will tentatively reconvene on Wednesday, June 13, 2018.

I. A regular meeting of the Elm Creek Watershed Management Commission was called to order at 11:40 a.m., Wednesday, April 11, 2018, in the Mayor’s Conference Room, Maple Grove City Hall, 12800 Arbor Lakes Parkway, Maple Grove, MN, by Chairman Doug Baines.

Present were: Bill Walraven, Champlin; Sharon Meister, Corcoran; Doug Baines, Dayton; Joe Trainor, Maple Grove; Elizabeth Weir, Medina; Fred Moore, Plymouth; Kevin Jullie, Rogers; James Kujawa, Hennepin County Dept. of Environment and Energy (HCEE); Brian Vlach, Three Rivers Park District (TRPD); Jeff Weiss, Barr Engineering; and Judie Anderson and Amy Juntunen, JASS.

Also present: Todd Tuominen, Champlin; Kevin Mattson, Corcoran; Rick Lestina and Mark Lahtinen, Maple Grove; Catherine Cesnik and Ben Scharenbroich, Plymouth; and Andrew Simmons, Rogers.

A. Motion by Trainor, second by Walraven to approve the revised agenda.* *Motion carried unanimously.*

B. Motion by Meister, second by Walraven to approve the minutes* of the March 14, 2018, regular meeting. *Motion carried unanimously.*

C. Motion by Moore, second by Walraven to approve the April Treasurer’s Report and Claims* totaling $267,085.58. *Motion carried unanimously.*

II. Open Forum.

III. Action Items.

A. Project Review 2018-008 Hayden Hills Golf Course, Dayton,* was removed from the action agenda.

B. Project Review 2018-010 Greenway West, Plymouth.* This 40.2 acre site consists of five rural single family lots west of Troy Lane about 1/4 mile south of CR 47. The applicant is proposing to develop the area in two phases into 78 single family residential lots. The development must comply with the Commission’s Stormwater Man-
DATE: November 7th, 2018

TO: Elm Creek Watershed Management Organization

FROM: Kirsten Barta, Hennepin County Department of Environment and Energy

RE: Subwatershed assessment cost share policy recommendations

Below are the recommendations of the Hennepin County staff regarding the subwatershed assessment cost share policy as requested by the commissioners of the Elm Creek Watershed Commission.

1. Under item c of the subwatershed assessment portion of the cost share policy, it is recommended that some more clarification be added, for example:

   “Undertaken at the discretion of the Commission based on the information provided by cities in the completed SWA cost share application form”

2. Staff propose the following timeline for evaluating and executing SWA projects:

   January 15th – applications are due from cities

   February TAC meeting – Technical staff will have reviewed applications and prepared recommendations for the consideration of the TAC to be brought to the February Commission regular meeting.

   March – Budget work

   March/April following year – SWA delivered to commission

   August – BWSR grant applications due for implementation funding

3. The below criteria are suggested for evaluating the applications:

   a. Subwatershed is identified in the MPCA WRAPS or TMDL report as a priority
   b. Sponsor city shows active staff and financial support for implementation of projects identified within the SWA
   c. Sponsor city has the ability to leverage outside funding for implementation
Elm Creek Watershed Commission
Subwatershed Assessment Cost Share Application

Date:
Waterbody to be assessed:
Sponsor City:

Total cost estimate:
Anticipated City Contribution:
Anticipated Commission Contribution:
Firm(s) solicited:

**Background information**

Why is the sponsoring city interested in this SWA?

Other supporting documents showing water quality issues? Ex: TMDL, Stressor ID report, etc. Please provide web links

Any additional local knowledge of issues?

**Implementation**

What implementation support will the sponsoring city provide? Ex: funding, staff time, outreach, submitting a Clean Water Fund app, etc

Does the sponsoring city presently have plans to incorporate the SWA information into their planning or other work?

**Other information**

Is there anything else the Commission should know about the proposed SWA?

**Attachments**

Please attach a map of the proposed project area as well as any cost estimates solicited

SWA cost share application – 11/7/18 draft
Elm Creek Watershed Management Commission
Cost Share Policy

To facilitate implementation of improvement projects within the watershed, the Elm Creek Watershed Management Commission's Joint Powers Agreement (JPA) and Section V of its Second Generation Watershed Management Plan provide for a Capital Improvement Program (CIP). The JPA also describes how the costs of capital projects shall be allocated.

The Management Plan proposes to share the cost of high-priority watershed capital improvements and demonstration projects through the CIP. High-priority watershed capital improvements are those activities that go above and beyond general city management activities and are intended to provide a significant improvement to the water resources in the watershed. To be considered for inclusion in the CIP, projects must be identified in a Commission-adopted management plan, approved TMDL, or member local stormwater plan or CIP.

In order to identify projects for inclusion on its Capital Improvement Program, the Elm Creek Watershed Management Commission will accept city proposals for cost-share projects until March 15 of every year. Following that date, the Commission’s Technical Advisory Committee will review and score the submittals and make a recommendation regarding additions and revisions to the Commission’s existing CIP at their regular May meeting.

The Commission has developed a set of criteria by which proposed projects will be scored, with those projects scoring a certain minimum number of points on the submittal form screening questions advancing to a prioritization stage. (Refer to the Commission’s Capital Improvement Program Standards and Guidelines.)

Prior to consideration for funding, a feasibility study or engineering report must be written for the proposed project. The city acting as the lead agency for a proposed project will be responsible for the development of and the costs associated with the feasibility study/engineering report.

The Commission has elected to fund capital projects through an ad valorem tax levy. Under the authority provided by MN Stat 103B.251, Subd. 5, the Commission has the authority to certify for payment by the county all or part of the cost of an approved capital improvement. The Commission will pay up to 25 percent of the cost of qualifying projects. This amount will be shared by all taxpayers in the watershed, with the balance of the project cost being shared by the local government(s) participating in or benefiting from the improvement.

a. The Commission’s maximum annual share of an approved project is up to $250,000.
   1) The Commission’s share will be funded through the ad valorem tax levy – spread across all taxpayers within the watershed.
   2) The Commission will use a maximum annual levy of $500,000 as a working guideline.

b. The cities’ share will be a minimum of 75% of the cost of the project. The basis of this apportionment will likely be unique to each project. The 75% share will be apportioned to the cities in the following manner or in some other manner acceptable to them. For example,
   1) The area directly benefitting from the project will be apportioned 25% of the cost of the project. This will be apportioned to cities based on the proportion of lake or stream frontage.
2) 50% of the cost of the project will be apportioned based on contributing/benefiting area.

c. The cities will each decide the funding mechanism that is best suited to them for payment of their share, for example through special assessments, storm drainage utility, general tax levy, or watershed management taxing district.

d. Funding from grant sources may also be used to help pay the costs of the capital projects.

The Elm Creek Watershed Management Commission may consider Commission- or City-generated requests to undertake subwatershed assessments (SWAs). Primarily, SWAS will be completed in rural areas suspected of being high-nutrient loading and will be specific enough to identify potential load-reducing projects. SWAs will be

a. Identified in areas outside of the Municipal Urban Service Area (MUSA).

b. Supported by the City in which the SWA is located.

c. Undertaken at the discretion of the Commission.

d. Funded by a $15,000 maximum cap (grant or Commission funding) and a 20% match by the City requesting the SWA.
Jim, I spoke with Jennie Skanke (DNR Hydro southern metro) and her opinion was the same as mine that discharging ground water into a wetland would not negatively affect the wetlands ecology, chemistry, biota, etc. Many wetlands are groundwater fed so it should be a non-issue. Her stance from the DNR is they might be willing to write an email or letter stating this but would be reluctant to attend a meeting. I can discuss with Eric Mohring at our office too, he just came back from retirement.

Ben

Thanks Ben, I will do the same.

James C. Kujawa
Hennepin County Public Works
Department of Environment and Energy
701 Fourth Avenue South, Suite 700
Minneapolis, MN  55415
Direct Phone: 612-348-7338
Email: james.kujawa@hennepin.us

Boy, that’s a good question. We don’t deal with groundwater much here... Possibly Eric Mohring, our hydrologist but he’s only part time and has limited hours. I wonder if someone at the DNR might know more about this, possibly Jason Spiegel? I can ask around.

Ben
Ben. I was thinking of maybe having someone talk to this issue at a future TAC meeting of the ECWMC. Would you have anyone that you could recommend? I am not sure if it would be a biologist or hydrologist? Or who? But just some information on how they feel the well water could impact the wetlands biota and wetland ecology?

Or maybe you know a good source of information that I can access and provide to the TAC?

Thanks
Jim

James C. Kujawa
Hennepin County Public Works
Department of Environment and Energy
701 Fourth Avenue South, Suite 700
Minneapolis, MN  55415
Direct Phone: 612-348-7338
Email: james.kujawa@hennepin.us

---

**From:** Carlson, Ben (BWSR) <ben.carlson@state.mn.us>
**Sent:** Wednesday, October 17, 2018 11:21 PM
**To:** James C Kujawa <James.Kujawa@hennepin.us>
**Subject:** RE: [External] using wetlands as irrigation ponds

Agreed, her comments are valid but don’t necessarily relate to WCA. Thanks for the information Jim!

---

**From:** James C Kujawa [mailto:James.Kujawa@hennepin.us]
**Sent:** Friday, October 12, 2018 3:16 PM
**To:** Carlson, Ben (BWSR) <ben.carlson@state.mn.us>; Stacey L Lijewski <Stacey.Lijewski@hennepin.us>
**Subject:** FW: [External] using wetlands as irrigation ponds

FYI.......Please see Liz Weir’s comments regarding the irrigation wetland on the Encore site in Corcoran below. The Commission approved (by a unanimous vote) the replacement plan at their meeting on Wednesday. It was approved with the NWL on the irrigation pond set at 928.25. From a wetland value standpoint, she probably has some good points, but ultimately, I believe the TEP’s recommendations were sound as they relate to the WCA rules.

James C. Kujawa
Hennepin County Public Works
Department of Environment and Energy
701 Fourth Avenue South, Suite 700
Minneapolis, MN  55415
Direct Phone: 612-348-7338
Email: james.kujawa@hennepin.us
Dear Jim,

I disagree with the use of a wetland for irrigation. In a severe drought, the wetland would simply become a pass-through for ground water irrigation. That might keep up the water level in the wetland, but it's essentially using precious ground water to irrigate lawns, and ground water levels are dropping faster than replenishment. Also, how does the chemistry of ground water differ from the natural water within a wetland. How might it effect the wetland biota and wetland ecology?

I fear that we set an unfortunate precedent in passing the Corcoran Project, #2018-32W. I would hope that the Technical Advisory Panel resists any future efforts by developers to follow this path.

My best, Liz

***CAUTION: This email was sent from outside of Hennepin County. Unless you recognize the sender and know the content, do not click links or open attachments.***
Hi Jim,

I wanted to follow up regarding your interest in iron in wetland systems. I have attached two excerpts that may assist you in evaluating the HOA proposal. Let me know if you would like any of the citations found in either resource.

After our conversation it occurred to me that your description of using iron enriched groundwater to supplement wetland water withdraws and its potential impact on factors underpinning wetland ecological expression, in some respects, has a natural analog. Under common natural conditions, anoxic groundwater contains chemically reduced, water-soluble ferrous iron. When groundwater seeps to the oxygen-rich ground surface, it becomes chemically oxidized by bacteria. This results in iron becoming insoluble and depositing as ferric iron on the water surface as a red sheen or to the soil surface under drawdown conditions. This is the science underpinning the iron deposit (B6) indicator of wetland hydrology. I have observed this phenomena in fresh meadow and shallow/deep marsh environments with no visible stress to the wetland community.

However, as we know, chemical and biogeochemical processes are limiting and proceed under specific environmental conditions. The described process is thought to precipitate as ferric iron deposits, without the production of acid, provided water chemistry is sufficiently alkaline/geological materials act as agents of chemical buffering. The attached will expand on this and more.

I hope this information gives you a bit more to chew on. Sounds like an interesting proposal.

Alex Yellick

***CAUTION: This email was sent from outside of Hennepin County. Unless you recognize the sender and know the content, do not click links or open attachments.***
the numbers in this hypothetical example, it is easily seen that if the wetland is effective, and there is elevated metal in the incoming water, it will be difficult to meet stringent sediment quality standards (Kadlec, 1998).

11.6 THE OXIDE FORMERS

Iron
Iron is a metal that may occur at trace to high concentrations in wetland surface waters and sediments. It is required by plants and animals at significant concentrations. In plants, iron is an essential element in chlorophyll synthesis, cytochromes, and in the enzyme nitrogenase. In animals, iron is important in oxidative metabolism and is a key component in hemoglobin.

Iron at low to moderate concentrations is not generally regarded as a threat to human health or aquatic life. The U.S. EPA has recommended a continuous concentration criterion of 1,000 µg/L for protection of freshwater aquatic environments, and a drinking water health criterion of 300 µg/L (U.S. EPA, 2002b). The province of Ontario, Canada, has a lower standard for protection of aquatic life, also 300 µg/L. Perhaps the greater concern is for the blanketing effect of thick deposits of iron precipitates in wetlands designed to treat high iron concentrations (Kelly-Hooper, 1999).

High concentrations of soluble iron in surface water and wetland systems may result from natural or artificial iron sources, typically as seeps of ferrous iron and iron sulfides (pyrites) from anaerobic groundwaters. Iron bacteria that produce ocher, such as *Leptothrix ochracea* and *Spirothrix ferrugineum*, derive their energy needs from the oxidation of reduced iron. These bacteria typically occur in wetland areas where anoxic waters meet aerated surface conditions, such as upwelling springs or other venting groundwaters. At such locations, reddish brown flocculent deposits form.

Iron in the wetland waters may be dissolved or particulate. Most reported wetland studies do not specify which forms were determined. This may be a critical unresolved issue, because there may be very large amounts of suspended iron in wetland waters. Gammons et al. (2000b) report that the iron concentrations between filtered and unfiltered samples can differ by a factor of up to 100. Their data may be approximated as unfiltered iron equal to the square root of filtered iron, over the range 10–5,000 µg/L.

**Wetland Storage and Processing of Iron**

In an oxygenated environment, ferric iron is present as insoluble oxyhydroxides, denoted as FeOOH. If there is not sufficient alkalinity in the water, the reaction produces acidity:

\[
\text{Fe}^{3+} + \frac{1}{4} \text{O}_2 + \text{H}^+ \rightarrow \text{Fe}^{3+} + \frac{1}{2} \text{H}_2\text{O} \quad (11.24)
\]

\[
\text{Fe}^{3+} + 2\text{H}_2\text{O} \rightarrow \text{FeOOH}^{(s)} + 3\text{H}^+ \quad (11.25)
\]

However, if there is sufficient alkalinity, removal of iron to precipitates is not accompanied by a decrease in pH:

\[
\text{Fe}^{3+} + \frac{1}{4} \text{O}_2 + 2\text{HCO}_3^- \rightarrow \text{FeOOH} + 2\text{CO}_2 + \frac{1}{2} \text{H}_2\text{O} \quad (11.26)
\]

The water is deemed acidic for iron removal if the ratio of iron to CaCO₃ alkalinity is greater than 1.1 (Younger et al., 2002).

Oxidation and reduction of iron occurs relatively easily depending on redox potential and pH (Faulkner and Richardson, 1989). Fe⁺₃, or ferric iron, is the dominant form under oxidized conditions (Eₚ > 0 at pH ≥ 6.5). Fe⁺₂, or ferrous iron, is the dominant form under reduced conditions in wetlands and other aquatic environments. Fe⁺₃ forms stable complexes with a variety of ligands. It joins with the hydroxide iron in surface waters to form reddish-brown ferric hydroxide (Fe(OH)₃), which is also known as ocher. Ocher is insoluble and either settles to the bottom sediments or remains in suspension, adsorbed to living and dead organic matter (see Figure 7.10). Other important compounds formed by ferric iron include ferric phosphate (FePO₄), iron-humate complexes, and ferric hydroxide-phosphate complexes.

Ferric iron is reduced to the ferrous form under anaerobic conditions. The ferrous iron is more soluble, resulting in the release of dissolved iron and associated anions such as phosphate from anaerobic sediments in wetlands. The formation of this soluble ferrous iron may be controlled somewhat by sulfide, which forms the relatively insoluble ferrous sulfide (FeS). Sulfide formation is written as:

\[
\text{Fe}^{3+} + \text{HS}^- \rightarrow \text{FeS} + \text{H}^+ \quad (11.27)
\]

The required HS⁻ is microbially generated, and occurs preferentially in organic environments by the reduction of sulfate (see Equations 11.1, 11.2, and 11.3).

The role of sulfate-reducing bacteria (SRB) in the cycling of iron and sulfur was studied in a young constructed wetland located in Kanata, Ontario, Canada (Fortin et al., 2000). Sediments and water samples were collected over the course of one year within each of three FWS cells. SRB populations were largest during the cold winter months, when the temperature of the water was 1°C. The presence of high-SRB populations also corresponded to highly anoxic conditions within the sediments and to a decrease of sulfate concentrations, suggesting that cold temperature did not affect the activity of SRB. The results indicated that iron and sulfur cycling in the constructed wetland was active throughout the year, especially in the cold winter months. This suggests that iron removal in wetlands can be effective in temperate climates, even though the temperature of the water decreases drastically during the winter.

**Soils and Sediments**

Wetland soils can contain large amounts of iron, especially when exposed to metalliferous waters (Table 11.16). On a dry basis, ferric oxide contains 70% iron by weight (700,000 mg/kg), and this represents an upper limit to sedimentary iron concentrations in oxic wetland waters. Iron sulfides contain 53% (FeS₂) and 66% (FeS) iron. Such mineral precipitates are diluted by newly formed organic materials in the wetland...
TABLE 11.16
Iron Content of Top Sediments in a Variety of Wetlands

<table>
<thead>
<tr>
<th>Location</th>
<th>Notes</th>
<th>Water Source</th>
<th>Iron (mg/kg)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Michigan</td>
<td>Fen</td>
<td>Natural</td>
<td>4,924</td>
<td>Faulkner and Richardson (1989)</td>
</tr>
<tr>
<td>North Carolina</td>
<td>Pocosin</td>
<td>Natural</td>
<td>2,370</td>
<td>Faulkner and Richardson (1989)</td>
</tr>
<tr>
<td>Maryland</td>
<td>Bog</td>
<td>Natural</td>
<td>5,710</td>
<td>Faulkner and Richardson (1989)</td>
</tr>
<tr>
<td>Maryland</td>
<td>Swamp</td>
<td>Natural</td>
<td>5,410</td>
<td>Faulkner and Richardson (1989)</td>
</tr>
<tr>
<td>North Carolina</td>
<td>Swamp</td>
<td>Natural</td>
<td>1,301</td>
<td>Faulkner and Richardson (1989)</td>
</tr>
<tr>
<td>Ireland</td>
<td>Salt marsh</td>
<td>Natural</td>
<td>6,000-38,000</td>
<td>Doyle and Otte (1997)</td>
</tr>
<tr>
<td>Germany</td>
<td>Lake Schloßsee</td>
<td>Natural</td>
<td>56,000</td>
<td>Wetzel (1975)</td>
</tr>
<tr>
<td>Maine, Ontario</td>
<td>Cattail marsh</td>
<td>Urban stormwater</td>
<td>12,000</td>
<td>Goulet and Pick (2001)</td>
</tr>
<tr>
<td>Maine, Ontario</td>
<td>Cattail marsh</td>
<td>Urban stormwater</td>
<td>2,500</td>
<td>Goulet and Pick (2001)</td>
</tr>
<tr>
<td>Maine, Ontario</td>
<td>Cattail marsh</td>
<td>Acid mine drainage</td>
<td>1,500</td>
<td>Goulet and Pick (2001)</td>
</tr>
<tr>
<td>Maine, Ontario</td>
<td>Cattail marsh</td>
<td>Tailings leachate</td>
<td>1,500</td>
<td>Goulet and Pick (2001)</td>
</tr>
<tr>
<td>Maine, Ontario</td>
<td>Cattail marsh</td>
<td>Tailings leachate</td>
<td>115,500</td>
<td>De Volder et al. (2003)</td>
</tr>
<tr>
<td>Maine, Ontario</td>
<td>Cattail marsh</td>
<td>Tailings leachate</td>
<td>60,000-85,000</td>
<td>Ye et al. (2001a,b)</td>
</tr>
<tr>
<td>Maine, Ontario</td>
<td>Ptain marsh</td>
<td>Municipal</td>
<td>30,575</td>
<td>NADB database (1998)</td>
</tr>
<tr>
<td>Maine, Ontario</td>
<td>Ptain marsh</td>
<td>Municipal</td>
<td>9,400</td>
<td>NADB database (1998)</td>
</tr>
<tr>
<td>Maine, Ontario</td>
<td>Ptain marsh</td>
<td>Pulp and paper</td>
<td>2,560-2,720</td>
<td>Eckhardt et al. (1997)</td>
</tr>
<tr>
<td>Poland</td>
<td>11 lakes</td>
<td>Brown coal pits</td>
<td>115-21,500</td>
<td>Samecka-Cymerman and Kemper (2001)</td>
</tr>
</tbody>
</table>

Environment, and lesser concentrations are observed. For instance, Doyle and Otte (1997) measured 6,000–40,000 mg/kg, and higher values in the rhizosphere and near worm burrow walls.

Freeze-coring and analysis of the wetland substrates indicated that total sulfur was present in three forms, in the following proportions (Younger, 2000): FeS: 35%; Fe2S3: 31%; S2: 34%.

On the basis of these observations, it was postulated that pH rise was due to the consumption of protons via reactions involving reduction of ferric hydroxide and precipitation of elemental sulfur. The removal of iron from solution and formation of significant quantities of S2 is consistent with the following coupled reactions:

\[
5H^+ + 2FeOOH + HS^- \rightarrow 2Fe^{2+} + S^2 + 4H_2O \quad (11.28)
\]

\[
Fe^{2+} + H_2S + 2HCO_3^- \rightarrow FeS + 2H_2O + 2CO_2 \quad (11.29)
\]

Goulet and Pick (2001) found that the presence of cattails had little effect on the partitioning of iron in shallow wetland sediments in FWS wetlands. Studies at four Ontario treatment wetlands showed total iron in the sediments of 2,000–12,000 mg/kg, with sediment organic content of 8–20%. About half of the sediment iron was in reactive forms, oxides, monosulfides, or sorbed on organic matter. The balance was dominated by forms associated with either the pyrite (one wetland) or the silicate fraction of the sediment (three wetlands).

Wetland Plants

Metals reach plants via their fine root structure, and most are intercepted there. Some small amounts may find their way to stems, leaves, and rhizomes. Upon root death, some fraction of the metal content may be permanently buried, but there are no data on metal release during root decomposition. However, wetland plants bring oxygen to their root zone to maintain respiration, and some fraction is lost by radial diffusion away from the roots. This creates small aerobic zones near the roots, in which iron precipitates may form. These are termed iron plaques.

Nonetheless, some iron is taken up into aboveground tissues. Iron occurs in wetland plants at concentrations ranging from about 200–2,000 mg/kg dry mass (Vymazal, 1995). Plant roots contain a much higher concentration of iron than stems or leaves (Table 11.17). Uptake by plants and algae may be for purposes of growth enhancement, or at higher metal concentrations for protective purposes. Biomagnification of iron does not occur.

A common concept of wetland treatment is the perceived risk of seasonal release of contaminants during winter, when wetland macrophytes die back. This theoretical risk was investigated experimentally in mesocosm experiments on plant litter collected from long-established mine water treatment wetlands in the United Kingdom (Batty and Younger, 2002). Metals were not released from the plant litter, and iron concentrations in the litter increased after 6 months of decomposition, which was attributed to adsorption. Field studies undertaken within the PIRAMID project (PIRAMID Consortium, 2003a) found that wetlands were net sinks for iron in all seasons.

Performance of Wetlands for Iron Removal

Wetlands interact strongly with iron in a number of ways, and thus are capable of significant metal removal. Three major mechanisms are operative:
### Table 11.16
Iron Content of Top Sediments in a Variety of Wetlands

<table>
<thead>
<tr>
<th>Location</th>
<th>Notes</th>
<th>Water Source</th>
<th>Iron (mg/kg)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Michigan</td>
<td></td>
<td>Natural</td>
<td>4,934</td>
<td>Fiskerich and Richardson (1989)</td>
</tr>
<tr>
<td>North Carolina</td>
<td>Fen</td>
<td>Natural</td>
<td>2,750</td>
<td>Fiskerich and Richardson (1989)</td>
</tr>
<tr>
<td>North Carolina</td>
<td>Bog</td>
<td>Natural</td>
<td>5,710</td>
<td>Fiskerich and Richardson (1989)</td>
</tr>
<tr>
<td>North Carolina</td>
<td>Swamp</td>
<td>Natural</td>
<td>5,492</td>
<td>Fiskerich and Richardson (1989)</td>
</tr>
<tr>
<td>North Carolina</td>
<td>Swamp</td>
<td>Natural</td>
<td>1,361</td>
<td>DeVolker and O’Toole (1997)</td>
</tr>
<tr>
<td>Ireland</td>
<td>Salt marsh</td>
<td>Natural</td>
<td>6,000-20,000</td>
<td>DeVolker and O’Toole (1997)</td>
</tr>
<tr>
<td>Germany</td>
<td>Lake Shinonoe</td>
<td>Natural</td>
<td>12,000</td>
<td>Goost and Pick (2001)</td>
</tr>
<tr>
<td>Pearl, Ontario</td>
<td>Canals</td>
<td>Natural</td>
<td>2,250</td>
<td>Goost and Pick (2001)</td>
</tr>
<tr>
<td>Missouri, Ontario</td>
<td>Canals</td>
<td>Natural</td>
<td>1,500</td>
<td>Goost and Pick (2001)</td>
</tr>
<tr>
<td>Wisconsin, Ohio</td>
<td>Canals</td>
<td>Natural</td>
<td>1,500</td>
<td>Goost and Pick (2001)</td>
</tr>
<tr>
<td>Indiana</td>
<td>Lake Michigan</td>
<td>Natural</td>
<td>4,500</td>
<td>DeVolker and O’Toole (2003)</td>
</tr>
<tr>
<td>West Virginia</td>
<td>Tailing blanket</td>
<td>Municipal</td>
<td>60,000-45,000</td>
<td>NARDB database (1998)</td>
</tr>
<tr>
<td>White River</td>
<td>Tailing blanket</td>
<td>Municipal</td>
<td>38,575</td>
<td>NARDB database (1998)</td>
</tr>
<tr>
<td>Pasture, Arkansas</td>
<td>Tailing blanket</td>
<td>Municipal</td>
<td>18,337</td>
<td>NARDB database (1998)</td>
</tr>
<tr>
<td>California</td>
<td>Tailing blanket</td>
<td>Municipal</td>
<td>17,699</td>
<td>NARDB database (1998)</td>
</tr>
<tr>
<td>Mexico, New York</td>
<td>Tailing blanket</td>
<td>Municipal</td>
<td>9,450</td>
<td>NARDB database (1998)</td>
</tr>
<tr>
<td>Poland</td>
<td>Tailing blanket</td>
<td>Municipal</td>
<td>2,650-2,750</td>
<td>Bledsoe et al. (1997)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>115-22,300</td>
<td>Sanecki-Cyzymer and Kempers (2001)</td>
</tr>
</tbody>
</table>

**Note:** FWS = free water surface; SSF = subsurface flow; CW = congested wetland; Nat = natural wetland

---

**Environmental iron concentrations are observed for instance.**

Dye and O’Toole (1997) measured 6,000-40,000 mg/kg, and higher values in the atmosphere and near storm borrow walls.

**Feozenation and analysis of the wetland substrates indicated that total sulfur was present in three forms, in the following proportions (Younger, 2000): FeS: 35%; Fe3S2: 2%; Fe2S3: 3%.**

On the basis of these observations, it was postulated that pH rise was due to the consumption of protons via reactions that cause iron sulfate at concentrations ranging from about 20-200 mg/kg dry mass (Vazzano, 1995). Plant roots contain a much higher concentration of iron than stems or leaves (Table 11.17). Upshots by the plant and algae may be consistent with growth enhancement or, at higher metal concentrations for protective purposes. Biomagnification of iron does not occur.

A common concept of wetland treatment is the perceived risk of seasonal release of contaminants during winter, when wetland macrophytes die back. This theoretical risk was investigated experimentally on a continuous treatment plant that was constructed from long-established mine waste treatment in the United Kingdom (Batty et al., 2002). Metals were not removed, but in the winter, leaf tissue content in the lake increased after 6 months of cold weather, which was attributed to adsorption. Field studies undertaken within the PIRAMID experimental (2003A) found that wetlands were not sites for iron in all seasons.

**Performance of Wetlands for Iron Removal**

Wetlands interact strongly with iron in a number of ways, and thus can be capable of significant metal removal. Three major mechanisms are operative:

1. **Binding to soils, sediments, particulates, and soluble organics**
2. **Precipitation as insoluble salts, predominantly sulfides and carbonates**
3. **Uptake by plants, including algae (however, after the plant senescence or algal die back most iron is leached out)**

**Information on the effects of wetlands on iron concentrations has been reported at a low level of detail, with emphasis on treatment systems, with few data sets that exist.**

**Example performances are shown in Table 11.18.**

**Coal Mine Drainage Wetlands**

Information on iron removal in wetlands is available primarily from acid mine drainage (AMD) wetland treatment systems in the United States (Griswold, 1987; Kaiser and Hedin, 1989; Hedin, 1989). However, treatment wetlands also become widely used in the United Kingdom in the 1990s. Younger (2000) lists 24 wetland systems, which fall into two principal categories:

- **Aerobic FWS wetlands**, usually vegetated by Phragmites australis. These are typically used for iron reduction in coal mine drainage and can be used to remove iron from AMD.
- **FWS with an anoxic compost substrate**, typically vegetated by Phragmites australis. Because of the organic substrate, these are sometimes referred to as anaerobic FWS wetlands. These are typically used for iron reduction in more acidic waters.

Of the 137 AMD wetland treatment sites reviewed by Younger (1989), 96% had influent iron concentrations less than 50 mg/L. An average total iron concentration of 60 mg/L was reduced to an average effluent concentration of 15 mg/L.
for an average iron removal efficiency of 58.2% and a median value of 80.9% (Winder, 1989). This median is clearly also close to that experienced at the U.K. treatment plants (Table 11.18). This treatment efficiency was given by Kitching and Hadin (1989).

Because the iron removal rate is correlated with iron loading rate (Hedin and Nisius, 1990), lower removal rates are expected at low influent concentrations.

Landfill Leachate Wetlands

Scrap iron items discarded to landfill create an underground source of iron, which then appears as a constituent of leachate. Raw leachate concentrations may be as high as 500 mg/L, mostly in soluble form because of the anaerobic condition in the pile (McKean and Rovers, 1999). At such high concentrations, iron precipitates pose a serious clogging threat even for FWS wetlands. Consequently, aeration and precipitation steps are often included in the treatment steps (Hoover et al., 1998; Loer et al., 1999). Oxidation/precipitation basins are open water impoundments designed to provide aeration for precipitation of soluble iron, detention time to settle precipitates, and storage volumes for accumulating precipitate sludge. These basins are a key component of the passive systems where iron is present. A detention time of at least 24 hours is recommended to produce a stable iron sludge (Hoover et al., 1998). The leachates in Table 11.18 discharge all had lower iron than that in raw leachates due to iron removal by precipitation, sedimentation, and precipitation.

FWS wetlands produce considerable further reduction in iron, with a median removal of 85% for the six systems in Table 11.18. Subsurface systems may also be used. For instance, Sartore et al. (1998) measured an iron removal efficiency of 78.5% in a FWS wetland planted with common reed (Phragmites australis). Belo et al. (1997) reported an iron removal of 85% in a landfill leachate HSSF system in Sweden with aerobic concentration of 30 mg/L.

Other Water Sources

A variety of other wastewaters have been subjected to wetland treatment, and in some cases the iron content of inflows and outflows has been measured. This is usually ancillary monitoring, which does not target regulatory criteria. Ryckewaert et al. (2003) found the iron reduction of 27% in a FWS-HSSF system in Uganda treating cobalt extraction wastes. The iron content of pretreated municipal wastewater is likely to be less than 1 mg/L in many regions, although higher in regions with high occurrence of iron minerals (Table 11.18). For example, Vymazal (2000) reported reduction of the iron from 11 to 0.72 mg/L in a HSSF system treating municipal sewage near Prague in the Czech Republic. However, other studies from the Czech Republic (Table 11.18) showed rather negative removal effects for HSSF systems due to conditions discussed above. When iron concentrations are very low, the use percent removal is no longer an appropriate measure, because very small excursions in concentration can greatly affect the percentage. Nonetheless, removals may or may not occur for very low iron content waters (see Table 11.18).

For the average iron removal efficiency of 58.2% and a median value of 80.9% (Winder, 1989). This median is clearly also close to that experienced at the U.K. treatment plants (Table 11.18). This treatment efficiency was given by Kitching and Hadin (1989).

Because the iron removal rate is correlated with iron loading rate (Hedin and Nisius, 1990), lower removal rates are expected at low influent concentrations.

Landfill Leachate Wetlands

Scrap iron items discarded to landfill create an underground source of iron, which then appears as a constituent of leachate. Raw leachate concentrations may be as high as 500 mg/L, mostly in soluble form because of the anaerobic condition in the pile (McKean and Rovers, 1999). At such high concentrations, iron precipitates pose a serious clogging threat even for FWS wetlands. Consequently, aeration and precipitation steps are often included in the treatment steps (Hoover et al., 1998; Loer et al., 1999). Oxidation/precipitation basins are open water impoundments designed to provide aeration for precipitation of soluble iron, detention time to settle precipitates, and storage volumes for accumulating precipitate sludge. These basins are a key component of the passive systems where iron is present. A detention time of at least 24 hours is recommended to produce a stable iron sludge (Hoover et al., 1998). The leachates in Table 11.18 discharge all had lower iron than that in raw leachates due to iron removal by precipitation, sedimentation, and precipitation.

FWS wetlands produce considerable further reduction in iron, with a median removal of 85% for the six systems in Table 11.18. Subsurface systems may also be used. For instance, Sartore et al. (1998) measured an iron removal efficiency of 78.5% in a FWS wetland planted with common reed (Phragmites australis). Belo et al. (1997) reported an iron removal of 85% in a landfill leachate HSSF system in Sweden with aerobic concentration of 30 mg/L.

Other Water Sources

A variety of other wastewaters have been subjected to wetland treatment, and in some cases the iron content of inflows and outflows has been measured. This is usually ancillary monitoring, which does not target regulatory criteria. Ryckewaert et al. (2003) found the iron reduction of 27% in a FWS-HSSF system in Uganda treating cobalt extraction wastes. The iron content of pretreated municipal wastewater is likely to be less than 1 mg/L in many regions, although higher in regions with high occurrence of iron minerals (Table 11.18). For example, Vymazal (2000) reported reduction of the iron from 11 to 0.72 mg/L in a HSSF system treating municipal sewage near Prague in the Czech Republic. However, other studies from the Czech Republic (Table 11.18) showed rather negative removal effects for HSSF systems due to conditions discussed above. When iron concentrations are very low, the use percent removal is no longer an appropriate measure, because very small excursions in concentration can greatly affect the percentage. Nonetheless, removals may or may not occur for very low iron content waters (see Table 11.18).

Examples and Models for Iron Removal in Treatment Wetlands

It is likely that wetland removal performance in FWS systems is area-specific (Younger et al., 2003), rather than volum-specific. Three simple predictive approaches have been suggested:

- Zero-order removal. Specification of a supposedly constant area removal rate, in kg ton. This corresponds to zero-order removal kinetics, for which removal is independent of the iron concentration.
- First-order removal. The reduction rate is proportional to the iron concentration (Tartis et al., 1999). Some of the contributing processes are first-order, such as oxidation of ferrous iron and bacterial sulfate reduction (Younger et al., 2005), as is the process of particular settling of precipitates. However, global removal does not necessarily follow such a model.

Data from 35 FWS natural wetlands receiving inputs of ferrous acid mine drainage to western Pennsylvania served as a basis for evaluating the merits of these three approaches.

Stark and Williams (1995) found that percentage removal was a better index of treatment than an areal removal. Nevertheless, a fixed areal removal rate of 10 kg of Fe per ha is accepted as adequate design guidelines for high levels of removal, while 20 kg of Fe per ha is allowable to cause considerable improvement (Heinri et al., 1994; PIRAMID Consortium, 2002).

Tartis et al. (1999) further analyzed the 35-wetland data set, and concluded that percentage removal should not be used, and that areal removal did not separate the effects of iron concentration and flow rate. They recommended the first-order model (Equation 11.37), and found a median reaction constant of 0.18 m/s (66 mm/s) for calibrations of a plug flow version of the model.

Maiden et al. (1997) performed a factorial experiment varying flow rate and inlet concentrations to wetland mesocosms. They found that outlet iron concentrations increased with increasing inlet iron loadings. Analysis of their data produces a rate constant of 42 m/s according to Equation 11.17.

More complex models have been proposed, but these have not yet reached a point of usefulness in wetland design. For instance, the proposed model of Flanagan et al. (1994) presupposes that removal is to sulfides, and contains no sedimentation step for ferric oxyhydroxides. Modeling software has been developed allowing simulation wetland systems (NOAH2D (PIRAMID Consortium, 2003b). NOAH2D simulates overland flow and solute transport through reed beds, allowing for fractional resistance to flow offered by wetland vegetation, and also allows for exchange with (and hyporheic flow within) permeable wetland bed sediments. This code is a research tool, rather than software available for design. The complexity of the NOAH2D code requires very long run times, and full calibration is acknowledged as unlikely by the authors (Younger et al., 2002). The Manyan et al. (1997) findings suggest the use of a graphical representation of wetland efficient iron versus inlet iron loading. These variables are completely independent, and avoid the artifact of spurious correlations caused by the inclusion of an order of magnitude in both the x and y axes. Information from 46 constructed wetlands was used to form such a
display (Figure 11.12), together with the many others at (1997) monsoon season. There were samples for this graph: above about 100 g/m²-year loading, there is an increase in outlet iron with increased iron loading; below about 100 g/m²-year loading, there is a smaller but noticeable increase in outlet iron with increased iron loading. The 100 g/m²-year loading also represents the maximum that may be imposed without wet- land effluent concentrations exceeding 1 mg/L. The 1-2 valves previously cited were not determined or verified for the low loading region.

In the low loading region, behavior probably reflects localized recirculation of solids and from precipitated and adsorbed forms, in response to localized variations in pH and mediated by diffusion to and from the water column. Background levels may also result from suspended particulate forms of iron. Uptake into aboveground macrophyte plant parts is low, but larger amounts of iron are found in roots, so that overall plant cycling can be of importance. Batty and Youngh (1998) also found that dissolved iron concentrations in wetland waters were at or below 1 mg/L, direct uptake of iron by plants could account for 100% of iron removal. This finding explained why aerobic reed beds removed dissolved iron at far greater rates than would be anticipated on the basis of the first-order kinetics of Fe(VI) oxidation. Batty and Youngh (1998) also found that 1 mg/L iron was also a significant indicator of healthy growth of Phragmites australis; at greater concentrations, the plants were not productive, while at lower concentrations were healthy.

Variability of Iron Removal in Treatment Wetlands

Interstream variability in the Scandinavian and Williams (1995) data sets was relatively high, with removal of 464 ± 28% (mean ± SD). It was less in the Manglo et al. (1999) monsoon cone, with removal of 19% ± 3% (mean ± SD). The monsoon cones were operated in the laboratory, and had no variability in depth, aspect ratio, water conditions, or substrate, and were all operated at circumneutral pH.

Instream variability has not been reported for continuous flow wetlands, but it is high for event-driven wetlands. For example, the coefficient of variation of outlet iron from the Hidden River wetland in Florida was 47.2% (ASCN, 2003).

Summary

Iron is effectively removed by treatment wetlands over the high end of the concentration range, which is typical of acid mine drainage and landfill leachates. Rates are rapid, and significant loadings may be removed. At low concentrations, and low uptake rates, iron can be a minor cycling factor. In higher iron concentrations, loadings can be significant. Accumulation of these materials is a factor in the ecological status of the wetland.

ALUMINUM

Aluminum occurs naturally in surface waters, to a small extent in the hydrated ironic oxide, and to a greater extent complexed with silicates in a colloidal form. Aluminum solubility varies with pH. It is least soluble at pH 7 and increases in solubility as Al³⁺, Al(OH)³⁺, and Al(OH)₂⁺ ions at 4 < pH < 5.5; and as aluminum ion (Al³⁺) at pH > 5.2. Solvating water and Phye, 1995. Aluminum precipitates as amorphous Al(OH)₃, which may slowly form the mineral Gibbsite (Barkworth et al., 2005). Precipitation rate is natural in wetland waters, but is of interest in treatment processes when there is a need to add aluminum coagulants or salts for purposes of phosphorus removal.

Wetland Processing and Storage of Aluminum

Aquatic systems typically contain low concentrations of total aluminum. In the Aridlands region of the eastern United States, 203 lakes had a mean pH of 6.3, and a mean total Al = 138 g/L. In Florida, 168 lakes had mean pH = 6.3 and a mean total Al = 89 g/L. Lakes are typically net sinks for aluminum, with 10–20% retention for low pH, and 70% for circumneutral conditions (Gensier and Psyte, 1999). Organic complexation occurs in natural waters, as binding to humic substances. Tannin cations such as Al³⁺ are more susceptible to binding to organic ligands than divalent cations. Peat based wetlands often provide the conditions of low pH that foster Al³⁺, and hence dissolved organic carbon, is an important factor in water aluminum water chemistry.

Aluminum is toxic to many species of algae, with effects being observed over a range of a few hundred to a few thousand mg/L total aluminum. Total dissolved aluminum is much less affected, and do not biomagnify aluminum. Wetland macrophytes are tolerant of high aluminum concentrations, in the thousands of mg/L total aluminum. Shallow bog fish are susceptible to a variety of effects for concentrations of a few hundred to a few thousand μg/L. U.S. EPA (1986a) ambient water quality standards are rated at 87 mg/L (chronic) and 40 mg/L (acute), but research is currently in progress to more accurately measure these numbers.

Aluminum uptake by emergent and floating plants (Typha and Lemna) was found mainly in the roots and rhizomes (Golterman, 2004; Golterman et al., 2004). Roots also had the highest aluminum concentrations at the lower flows. In wetlands. Separate studies (Table 11.13; Wiss, Gerla, and Associates, 2002). Sediments generated in treatment wetlands are often high in aluminum, with values in treatment wetlands ranging from 1.4% (Teas Rio, Arizona) to 4% (Sacramento, California) (Noble and Associates, 2003). The process of phosphorus adsorption onto aluminum hydroxides has seen extensive application to water treatment and lake metaphosphatation. However, aluminum hydroxides are controlled in excess dissolved phosphorus. Alum or aluminum chloride additions are designed to form a floc of inorganic aluminum (AlO₄), which in turn adsorbs phosphate, without coagulation, with the addition of a precipitating step for waters sent to wetlands for further polishing. Wetlands are also the recipients of water treatment backwash and sludge (Kugawa et al., 2001). Consequently aluminum is sometimes a contaminant in treatment wetland influents, either by intentional or accidental discharges.

Aluminum is a strong determinate of the phosphorus adsorption capacity in wetland soils (Reddy and D'Angelo, 1994). The phosphorus adsorption process in treatment wetlands has a strong mechanism, which is exhausted when all antecedent soil sorption sites are used. However, the amount of phosphorus that may be bound to soil wetlands forms an important part of the wetland start-up capacity for phosphorus removal. A phosphorus removal project has been implemented using aluminum slag alone in a wastewater treatment plant as a soil amendment (SAIC, 2005).

There are two main groups of projects that have measured aluminum removal in wetlands.

- FWS wetlands, which typically receive low concentrations and loads of aluminum
-SSF wetlands, which, for aluminum, typically treat acid mine waters with quite high concentrations and loads

Aluminum Removal Processes

Mine waters with pH less than 4 commonly contain high concentrations of aluminum (>50 mg/L) (Younger et al., 2002). Aluminum is present as Al³⁺ and, in the Al³⁺ and, the Al³⁺ and, Al³⁺ species as well. Al³⁺ + 3H₂O ↔ AlO(ОН)³⁺ + 3H⁺

Al⁢³⁺ + 3H₂O ↔ AlO(ОН)³⁺ + 3H⁺

The precipitate is amorphous, white, and of low density. It may later crystallize to gibbsite. These solids are common precipitates of many natural soils, and pose no problem as new sediment and soils (Younger et al., 2002). In the presence of dissolved sulfate, hydroxyaluminate precipitates may form as well.

Performance of Wetlands for Aluminum Removal

Examples of wetland reductions in concentration and mass removals are shown in Table 11.20. Performance is given across systems, with a median concentration reduction of 50%. Wieder (1989) examined 20 acid mine wetlands for reductions, and found a median of 47.8%, and a 5th percentile of 78.9%. Wieder (1989) also found one quarter of the acid mine wetlands had zero reduction. Vymazal and Krtaus (2003) reported decrease of aluminum concentration from 451 μg/L to less than 46 μg/L in a 62-m-long HSSF constructed wetland in the Czech Republic. The major decrease from 451 μg/L to 65 μg/L occurred within the first 15 m of the bed. Other results from HSSF wetlands treating municipal wastewater (Table 11.20) also showed a removal of 50% (50–52%).

Removal is to accretion in sediments, which form a source of potential return fluxes of aluminum, most likely as particulates. In a mesocosm study, Wieder et al. (1990) found that although aluminum was initially removed in a wetland, this process decreased with time, and the wetland began to export aluminum to downstream waters. Wieder et al. (1990) found that aluminum concentrations of 10 mg/L were found to be toxic to caracils, leading to their mortality and release of sorbed aluminum. Wieder et al. (1984) found that aluminum concentrations of 50 mg/L were found to be toxic to caracils, leading to their mortality and release of sorbed aluminum. Wieder et al. (1984) found that the increase in the use of aluminum as an additive for peat filters occurred from 3.25 to 13.634 mg/kg dry mass, with the majority bound as organic and oxide compounds.

Flanagan et al. (1994) proposed a model for wetland treatment of iron and aluminum in acid mine drainage, but its utility has apparently not been tested for aluminum (Müller and Witte, 1998). Regarding aluminum, Younger et al. (2002) conclude that much remains to be investigated in detail.

Example Treatment Wetlands for Aluminum Removal

There are few data sets on the removal of aluminum from types of incoming water other than acid mine drainage (Table 11.20). Backlund and Richardson (1999) conducted field mesocosm studies on aluminum dosing to foster phosphorus removal in treatment wetlands in south Florida. The "pre flows" generated did not settle effectively. Another FWS wetland study tested the concept of phosphorus removal via aluminum dosing of agricultural runoff, but with coagulation before sending the
Table 11.20: Examples of Aluminum Removal in Treatment Wetlands

<table>
<thead>
<tr>
<th>System</th>
<th>Water Source</th>
<th>Coagulant</th>
<th>Coagulant Addition</th>
<th>Coagulant Addition</th>
<th>Coagulant Addition</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wetland</td>
<td>Method</td>
<td>Concentration</td>
<td>Dose (g/L)</td>
<td>Dose (g/L)</td>
<td>Dose (g/L)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>ppm</td>
<td>(mg/L)</td>
<td>(mg/L)</td>
<td>(mg/L)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>3.8</td>
<td>0.4</td>
<td>1.1</td>
<td>3.8</td>
</tr>
<tr>
<td></td>
<td></td>
<td>3.5</td>
<td>0.5</td>
<td>0.9</td>
<td>3.5</td>
</tr>
<tr>
<td></td>
<td></td>
<td>3.0</td>
<td>0.6</td>
<td>0.4</td>
<td>3.0</td>
</tr>
<tr>
<td></td>
<td></td>
<td>2.5</td>
<td>0.7</td>
<td>0.3</td>
<td>2.5</td>
</tr>
<tr>
<td></td>
<td></td>
<td>2.0</td>
<td>0.8</td>
<td>0.2</td>
<td>2.0</td>
</tr>
<tr>
<td></td>
<td></td>
<td>1.5</td>
<td>0.9</td>
<td>0.1</td>
<td>1.5</td>
</tr>
<tr>
<td></td>
<td></td>
<td>1.0</td>
<td>1.0</td>
<td>0.0</td>
<td>1.0</td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.5</td>
<td>1.1</td>
<td>0.0</td>
<td>0.5</td>
</tr>
</tbody>
</table>

Manganese

Manganese is an essential element that is chemically similar to iron in its behavior in surface waters. Manganese is vital to plant photosynthesis and is used as an enzyme catalyst for the uptake and storage of carbon dioxide by plants and animals. Although manganese is toxic to some species at elevated concentrations, this situation occurs infrequently, typically with mining activities. Manganese concentrations greater than 2 mg/L were found to be toxic to laboratory experiments (Goldman and Horn, 1983). Manganese is not observed as a contaminant in wetlands receiving sewage.

Manganese may exist in oxidation states ranging from +4 to +7, but the manganese (+4) and manganic (+6) forms are most important in aquatic systems (Wetzel, 1983). At low redox potentials and low pH the predominant form is Mn(II) (Figure 11.13). The aquatic forms are pyrophosphate (MnPO₄) and humic acid (Mn₃O₄). Manganese and manganous forms are potentially toxic to aquatic life, and therefore manganese removal is an important consideration.

Wetland Processing and Storage of Manganese

In an oxic environment, manganese is primarily removed from solution by oxidation and hydrolysis (Sibora et al., 2000).

**FIGURE 11.13** Approximate distribution of manganese species according to pH and Eh. (Adapted from Wetzel, 1983, Ecological Aspects of Aquatic Systems, 2nd ed. Academic Press, New York.)

Mn⁺ transition to Mn(II) is slow and the process is generally considered to be biologically mediated, by bacteria, fungi, and algae. Oxidation to Mn(II) is intermediate, but likely as a substrate for other oxidants such as Mn(IV) or Mn(III) oxides, rather than the mineral manganite (Younger et al., 2002). Because manganese is soluble at acidic pH, there is a possibility for manganese precipitation in acidic waters. A result, most of the manganese is extractable by weak acids (e.g., 85%–95% after 24 h). Manganese oxidation may be inhibited in the presence of large amounts of iron, because iron exerts a preferential oxidation of manganese (Hedin and Nairn, 1993).

Mn(II) oxidation to Mn(IV) involves a series of redox reactions:

\[ \text{Mn}^{2+} + \frac{1}{2} \text{O}_2 + \text{H}_2 \text{O} \rightarrow \text{Mn}^{4+} + 2 \text{H}^+ \]  

(11.31)

However, dissolved Mn(II) can be removed by adsorption to the iron hydroxide floc.

Under anaerobic conditions, manganese is typically quite insoluble. Manganese sulfide (MnS) is stable at high pH and high concentrations of Mn(II). However, the carbonate has been precipitated for anaerobic removal in some wetland systems (MnCO₃), but pH must be above neutral for this process (Figure 11.13) (Wildeman et al., 1998a).
A Freundlich isotherm was fit to sorption data for river gravel and limestone substrates (Sikora et al., 2000). They reported $n = 0.43$ and $K = 22.6$ for river gravel, and $n = 0.43$ and $K = 6.6$ (see Equation 13.3). Manganese is found in wetland plants, algae, and sediments. Concentrations in sediments exposed to mine waters may be very high, up to 10,000 µg/L. For municipal wastewater treatment plants, sediment manganese is typically 200-500 µg/L (Table 11.21). Manganese concentrations in river water are of the same order of magnitude, and background values do not differ much from belowground values (Table 11.21).

The general conception is that it is the aerobic, surface waters of a wetland, oxidized forms are abundant, Mn(IV), while in the anoxic soils and sediments, reduced forms prevail, Mn(II) (LaForce et al., 2002). As a consequence, both manganese forms in sediments and mangnese cycling are driven by redox processes. For example, at the Cataldo, Idaho, mine drainage wetlands, the pattern of new aerated conditions during the summer. Samples used deeper waters and anoxic conditions in the sediments (LaForce et al., 2002). As a result, oxides were 54% of the total manganese in water, but only in spring. In general, the ratio of oxidized to reduced manganese species was 1:1 in spring and 1:8 in autumn.

The storage of manganese in wetlands entails little or no risk for waters other than acid mine drainages, because sediment contaminant concentrations are typically quite high (see Table 11.21). It is possible to estimate the sediment concentrations created by sustained removal of manganese, in terms of the dilution of the manganese inorganic, by the aceration of wetland sediments. For example, consider 500 dry g/m² of new sediments, originating from 1,000 dry g/m² of biomass production. If the sediment concentration is to be kept below some limiting level, even 500 µg/m² per year exceeds the level of the wetlands.

Performance of Wetlands for Manganese Removal

Manganese is typically removed in FWS wetlands (Table 11.22). For numerous systems, with inlet manganese ranging from 0.1-3800 µg/L, the median concentration reduction is 96% (Table 11.22). An increasing wetland exit concentration in response to increases in wetland manganese loading (Figure 11.14). Outlet concentrations are below 1.0 mg/L for wetlands receiving leachates and other non-alkaline drainage waters are much more heavily loaded, and can have exit concentrations up to 20 mg/L.

Surface horizontal subsurface detention wetlands, where filtration beds are usually anaerobic or anaerobic, may release manganese because the system is disturbed. However, the system is disturbed. In addition, manganese is often removed through the use of biological, chemical, and physical processes. For example, in a study at Nyack, Minnesota (1993) reported a substantial reduction of Mn in a HSSF constructed wetland in the Czech Republic. Average infiltration concentration of 278 mg/L was reduced to 55 µg/L. However, results from three other HSSF systems in the Czech Republic exhibit substantial Mn removal (Table 11.22). Vertical flow constructed wetlands due to higher oxygenation of the bed exhibit good removal of manganese (Table 11.23.

There are two principal methods of interpreting performance data for total removals in wetlands: the real load removal and the first-removal model. These are both explored in detail for iron and manganese in a 25-wetland data set by Tatums et al. (1999). The former presumes a fixed removal per square meter of wetland, regardless of inlet concentration. Since the latter incorporates the linear concentration and flow rate.

Problems of the zero-order model is that it cannot be used to estimate the removal rates on high loadings. The recent removal model is often used for low loadings. For example, the model describes a typical system described in Table 11.23. The vertical flow constructed wetland for higher oxygenation of the bed exhibit good removal of manganese (Table 11.23).

Example Treatment Wetlands for Manganese Removal

Quaking Houses, County Durham, United Kingdom (Younger et al., 2002). Lysimeter from an abandoned colliery, containing up to 15 mg/L manganese and 30 mg/L of iron, was discharging into the drainage wetlands over a one-year period. Plug flow rate constant averaged 1 m/day.

The Ye et al. (2000) Wild Creek, Idaho, study determined that Typha latifolia and Juncus effusus contained just 0.9% of the annual amount of manganese entering the system, with 99% of the added manganese in sediments and belowground plant parts in a HSSF wetland treating domestic wastewater, and 1.7% in aboveground plant parts. The majority of removed manganese is therefore associated with sediments, in sorbed or chemically precipitated forms. Long-term sustainable removal requires continuous maintenance of oxidizing conditions.

<table>
<thead>
<tr>
<th>Location</th>
<th>Wetland Plant</th>
<th>Water</th>
<th>Above</th>
<th>Below</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>TVA Mason Sholes, Alabama</td>
<td>SSF CW</td>
<td>Sphagnum</td>
<td>19</td>
<td>21</td>
<td>Bredov et al. (1998)</td>
</tr>
<tr>
<td>TVA Mason Sholes, Alabama</td>
<td>SSF CW</td>
<td>Sporobolus</td>
<td>10</td>
<td>17</td>
<td>Bredov et al. (1998)</td>
</tr>
<tr>
<td>TVA Mason Sholes, Alabama</td>
<td>SSF CW</td>
<td>Scirpus</td>
<td>11</td>
<td>14</td>
<td>Bredov et al. (1998)</td>
</tr>
<tr>
<td>TVA Mason Sholes, Alabama</td>
<td>SSF CW</td>
<td>Phragmites</td>
<td>10</td>
<td>36</td>
<td>Bredov et al. (1998)</td>
</tr>
<tr>
<td>TVA Mason Sholes, Alabama</td>
<td>SSF</td>
<td>Typha</td>
<td>7.600</td>
<td>7.697</td>
<td>200 (Ye et al. 2000, a)</td>
</tr>
<tr>
<td>TVA Mason Sholes, Alabama</td>
<td>FW</td>
<td>Typha</td>
<td>7.600</td>
<td>7.697</td>
<td>200 (Ye et al. 2000, a)</td>
</tr>
<tr>
<td>TVA Mason Sholes, Alabama</td>
<td>FW</td>
<td>Sporobolus</td>
<td>1.200</td>
<td>1.299</td>
<td>200 (Ye et al. 2000, a)</td>
</tr>
<tr>
<td>TVA Mason Sholes, Alabama</td>
<td>FW</td>
<td>Phragmites</td>
<td>3.000</td>
<td>3.120</td>
<td>200 (Ye et al. 2000, a)</td>
</tr>
<tr>
<td>TVA Mason Sholes, Alabama</td>
<td>FW</td>
<td>Typha</td>
<td>3.000</td>
<td>3.120</td>
<td>200 (Ye et al. 2000, a)</td>
</tr>
<tr>
<td>TVA Mason Sholes, Alabama</td>
<td>FW</td>
<td>Sphagnum</td>
<td>1.200</td>
<td>1.299</td>
<td>200 (Ye et al. 2000, a)</td>
</tr>
<tr>
<td>TVA Mason Sholes, Alabama</td>
<td>FW</td>
<td>Sporobolus</td>
<td>3.000</td>
<td>3.120</td>
<td>200 (Ye et al. 2000, a)</td>
</tr>
</tbody>
</table>
TABLE 11.23
Removal of Manganese in Constructed Wetlands

<table>
<thead>
<tr>
<th>System</th>
<th>Location</th>
<th>Water Source</th>
<th>Inlet (mg/L)</th>
<th>Outlet (mg/L)</th>
<th>Reduction (%)</th>
<th>Removal (g/m²/yr)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>FWS</td>
<td>United Kingdom</td>
<td>Coal mine water</td>
<td>19.0</td>
<td>16.0</td>
<td>15</td>
<td>158</td>
<td>Younger (2000)</td>
</tr>
<tr>
<td>35 systems</td>
<td>Wyoming, Pennsylvania</td>
<td>Coal mine water</td>
<td>16.0</td>
<td>8.4</td>
<td>46</td>
<td>366</td>
<td>Tarris et al. (1990)</td>
</tr>
<tr>
<td>10 systems</td>
<td>TVA (Eastern United States)</td>
<td>Coal mine water</td>
<td>9.4</td>
<td>3.0</td>
<td>67</td>
<td>133</td>
<td>Bokil (1992)</td>
</tr>
<tr>
<td>124 systems</td>
<td>Eastern United States</td>
<td>Coal acid mine</td>
<td>37.7</td>
<td>24.0</td>
<td>34</td>
<td>97</td>
<td>Wiede (1999)</td>
</tr>
<tr>
<td>Artesian</td>
<td>Pennsylvania</td>
<td>Coal acid mine</td>
<td>8.5</td>
<td>3.3</td>
<td>87</td>
<td>197</td>
<td>Heuer et al. (1998)</td>
</tr>
<tr>
<td>Yorkshire</td>
<td>Maryland</td>
<td>Coal acid mine</td>
<td>8.5</td>
<td>3.0</td>
<td>63</td>
<td>99</td>
<td>Wiede et al. (1988)</td>
</tr>
<tr>
<td>Clear Creek</td>
<td>Colorado</td>
<td>Coal acid mine</td>
<td>2.7</td>
<td>0.2</td>
<td>92</td>
<td>98</td>
<td>Wiede et al. (1988)</td>
</tr>
<tr>
<td>Massanutten</td>
<td>Virginia</td>
<td>Coal acid mine</td>
<td>9.3</td>
<td>4.0</td>
<td>56</td>
<td>135</td>
<td>Wiede et al. (1988)</td>
</tr>
<tr>
<td>6 Systems</td>
<td>Champion, Florida</td>
<td>Metal mine water</td>
<td>10.0</td>
<td>6.0</td>
<td>38</td>
<td>35</td>
<td>Wiede et al. (1988)</td>
</tr>
<tr>
<td>Escambia County</td>
<td>Florida</td>
<td>Metal mine water</td>
<td>0.005</td>
<td>0.001</td>
<td>96</td>
<td>124</td>
<td>Wiede et al. (1988)</td>
</tr>
<tr>
<td>New Hanover County</td>
<td>North Carolina</td>
<td>Metal mine water</td>
<td>0.001</td>
<td>0.00001</td>
<td>99</td>
<td>99</td>
<td>Wiede et al. (1988)</td>
</tr>
<tr>
<td>Inistiogearr</td>
<td>West Virginia</td>
<td>Metal mine water</td>
<td>0.002</td>
<td>0.0001</td>
<td>99</td>
<td>99</td>
<td>Wiede et al. (1988)</td>
</tr>
<tr>
<td>Norfolk</td>
<td>Massachusetts</td>
<td>Metal mine water</td>
<td>0.01</td>
<td>0.0001</td>
<td>99</td>
<td>99</td>
<td>Wiede et al. (1988)</td>
</tr>
<tr>
<td>Kedron</td>
<td>West Virginia</td>
<td>Metal mine water</td>
<td>0.001</td>
<td>0.0001</td>
<td>99</td>
<td>99</td>
<td>Wiede et al. (1988)</td>
</tr>
<tr>
<td>Section</td>
<td>Missouri</td>
<td>Metal mine water</td>
<td>0.001</td>
<td>0.0001</td>
<td>99</td>
<td>99</td>
<td>Wiede et al. (1988)</td>
</tr>
<tr>
<td>Median</td>
<td></td>
<td></td>
<td>4.7</td>
<td>0.5</td>
<td>98</td>
<td>98</td>
<td>Wiede et al. (1988)</td>
</tr>
</tbody>
</table>

FIGURE 11.15 Three types of constructed wetlands used in acid mine drainage treatment.

a stream (Stanley Burn), where it killed all aquatic life in its path. A 45-m² pilot wetland was found to be effective, comprising of a shallow pond with a substrate of stable waste (manure and straw). Subsequently, a full-scale, four-cell FWS series wetland was implemented in 1997. The first two cells, totaling 440 m², were the heart of the system, with two follow-on cells regarded as partly cosmetic. The two working cells contained baffles and islands to prevent short-circuiting. The full-scale system substrate was a mixture of municipal compost and manure. The average hydraulic loading was about 20 cm/d. The reduction in manganese was 26.4% over the first 27 months of operation, from 4.4 to 3.2 mg/L. The areal removal rate was 0.26 g/m²/d (950 kg/ha/yr), and the first-order areal removal rate constant was 0.22 m² (1.06 m).
Layer, in the oxidized rhizosphere of the plants, and on the leaf and stem surfaces of plants. Bacterial nitrogen fixation can be carried out by endosymbiotic bacteria of the genus *Rhizobium*, or by certain actinomycetes. Bacterial fixation is the most significant pathway for nitrogen fixation in salt marsh soils, while nitrogen-fixing bacteria are virtually absent from the low-pH peat of northern bogs. Cyanobacteria (blue-green algae) are common nitrogen fixing microorganisms in wetlands, occurring in flooded delta soils in Louisiana, in northern bogs, and in rice cultures.

**Dissimilatory Nitrate Reduction to Ammonia (DNRA)**

Because conversion of nitrate-nitrogen to diatomic and nitrous oxide is considered to be the primary transformation of nitrates in anoxicic soils, an additional process whereby nitrite-nitrogen is transformed in anoxicic conditions is often overlooked (Megoingal et al., 2004). The process—called dissimilatory nitrate reduction to ammonia (DNRA)—occurs as follows, with mobile nitrites as the initial form of nitrogen and less mobile ammonium as the product.

\[
4\text{NO}_3^- + 4\text{H}_2 + 2\text{H}^+ \rightarrow 3\text{H}_2\text{O} + \text{NH}_4^+ \quad (5.15)
\]

The process yields energy to many microorganisms capable of carrying out this process. The bacteria can be anaerobic, aerobic, or facultative. In some cases, nitrate reduction can be a more significant pathway than the other dissimilatory nitrate loss—denitrification. Many studies have supported the concept that high availability of organic carbon and/or low nitrite concentrations favor DNRA over denitrification (Megoingal et al., 2004).

**Anammox**

A newly discovered nitrogen transformation in anaerobic conditions is called *anammox* (for anaerobic ammonium oxidation). The process involves nitrite-nitrogen (rather than nitrate-nitrogen as originally thought) as the oxidant.

\[
\text{NO}_2^- + \text{NH}_4^+ \rightarrow 2\text{H}_2\text{O} + \text{N}_2 \quad (5.16)
\]

A few studies have definitively determined the impotence of anammox in the cycling of nitrogen in natural or created wetlands, but it does appear that nitrogen may be more important in wetlands where denitrification is limited by lack of organic carbon (Megoingal et al., 2004).

**Iron and Manganese Transformations**

Below the reduction of nitrate on the redox potential scale comes the reduction of manganese and iron (see Eqs. 5.6 and 5.7). Iron and manganese are among the most abundant minerals on the Earth, which are found in wetlands primarily in their reduced forms (ferrous and manganous, respectively; Table 5.2), and both are more soluble and more readily available to organisms in these forms. Manganese is reduced slightly before iron on the redox scale, but otherwise it behaves similarly to iron. The direct involvement of bacteria in the reduction of MnO_2_ (Eq. 5.6) has been questioned by some researchers, although several experiments have shown the generation of energy by the bacterial reduction of oxidized manganese (Landwehr, 1990).
Iron can be oxidized from the ferrous to the insoluble ferric form by chemosynthetic bacteria in the presence of oxygen:

$$4Fe^{2+} + O_2(ads) + 4H^+ \rightarrow 4Fe^{3+} + 2H_2O \quad (5.17)$$

Although this reaction can occur nonbiologically at neutral or alkaline pH, microbial activity has been shown to accelerate ferrous iron oxidation by a factor of 10^6 in coal mine drainage water (Singer and Stumm, 1970). A similar type of bacterial process is believed to exist for manganese.

Iron bacteria are thought to be responsible for the oxidation to insoluble toxic iron compounds in soluble ferrous iron that originates in anoxic groundwater in iron in its reduced ferrous form can be used in the iron and steel industry. Iron in its reduced ferrous form can be oxidized to ferric in oxidized conditions [Fe(OH)3]. This appearance gives a red color or brown color in oxidized conditions [Fe(OH)3]. This appearance gives a reddish brown color in oxidized conditions [Fe(OH)3]. This appearance gives a reddish brown color in oxidized conditions [Fe(OH)3]. This appearance gives a reddish brown color in oxidized conditions [Fe(OH)3]. This appearance gives a reddish brown color in oxidized conditions [Fe(OH)3]. This appearance gives a reddish brown color in oxidized conditions [Fe(OH)3]. This appearance gives a reddish brown color in oxidized conditions [Fe(OH)3]. This appearance gives a reddish brown color in oxidized conditions [Fe(OH)3]. This appearance gives a reddish brown color in oxidized conditions [Fe(OH)3]. This appearance gives a reddish brown color in oxidized conditions [Fe(OH)3]. This appearance gives a reddish brown color in oxidized conditions [Fe(OH)3]. This appearance gives a reddish brown color in oxidized conditions [Fe(OH)3]. This appearance gives a reddish brown color in oxidized conditions [Fe(OH)3]. This appearance gives a reddish brown color in oxidized conditions [Fe(OH)3]. This appearance gives a reddish brown color in oxidized conditions [Fe(OH)3]. This appearance gives a reddish brown color in oxidized conditions [Fe(OH)3].

Wetland soils section earlier in this chapter.

Iron and manganes are in their reduced forms can reach toxic concentrations in wetland soils. Ferrous iron, diffusing to the surface of the roots of wetland plants on root systems can oxidize biologically by oxygen leaking from root cells, immobilizing phosphorus and causing roots with iron oxide, and causing a barrier to nutrient uptake.

**The Sulfur Cycle**

Sulfur, as the 16th most abundant element in the Earth’s surface, occurs in seven different states of oxidation in wetlands, and like nitrogen, it is transformed through several pathways that are mediated by microorganisms (Fig. 5.11). Sulfur is found in several pathways that are mediated by microorganisms (Fig. 5.11). Sulfur is found in several pathways that are mediated by microorganisms (Fig. 5.11). Sulfur is found in several pathways that are mediated by microorganisms (Fig. 5.11). Sulfur is found in several pathways that are mediated by microorganisms (Fig. 5.11). Sulfur is found in several pathways that are mediated by microorganisms (Fig. 5.11). Sulfur is found in several pathways that are mediated by microorganisms (Fig. 5.11).

**Sulfate Reduction**

Sulfate reduction can take place as **an oxic metallure** reduction in which certain metals reduce the anodic metal to a different state of oxidation. This reaction results in the reduction of sulfur in anoxic environments.

$$4H_2 + SO_4^{2-} \rightarrow H_2S + 2H_2O + 2O_2 \quad (5.18)$$

This reaction can occur over a wide range of pH, with the highest rates occurring near neutral pH. There have been a few measurements of the rate at which hydrogen sulfide is produced in and released from wetlands, and those measurements have ranged over several orders of magnitude. It can be safely generalized that saltwater wetlands have...
higher rates of sulfide emission per unit area than do freshwater wetlands where
sulfate ions are much less abundant (~2700 mg/L in sea water; ~10 mg/L in fresh
water). Sulfur can also be released to the atmosphere as organic sulfur compounds,
especially as dimethyl sulfide (DMS), (CH$_3$)$_2$S; this flux is thought by some to be
more important than H$_2$S emissions from some wetlands. The
Sulfide Oxidation
Sulfides can be oxidized by both chemosynthetic and photosynthetic microorganisms
in the aerobic zones of some wetland soils. Certain species of Thiobacillus—and other bacteria collectively referred to as thiothrophic bacteria (formerly known as sulfur bacteria)—obtain energy from the oxidation of hydrogen sulfide to
sulfuric acid. The genus Thiobacillus includes several species that can further oxidize elemental sulfur to sulfate. These reactions are limited in the absence of oxygen.

\[
2H_2S + O_2 \rightarrow 2S + 2H_2O + \text{energy}
\]

and

\[
2S + 3O_2 + 2H_2O \rightarrow 2H_2SO_4 + \text{energy}
\]

Under anaerobic conditions, nitrate-nitrogen can be used as the terminal electron
corresponding to oxidizing hydrogen sulfides.

Photosynthetic bacteria, such as the purple sulfur bacteria (PSB) found in the
marshes and mud flats, are capable of producing organic matter in the presence of light according to the following equation:

\[
CO_2 + H_2S + \text{light} \rightarrow CH_2O + S
\]

This reaction uses hydrogen sulfide as an electron donor rather than H$_2$O, but is
otherwise similar to the more traditional photosynthesis equation. This reaction often
takes place under anaerobic conditions where hydrogen sulfide is abundant, but at
the surface of sediments where sunlight is also available.

Sulfide Toxicity
Hydrogen sulfide, which is characteristic of anaerobic wetland sediments, can be
toxic to rooted higher plants and microbes, especially in shallow waters where
the concentration of sulfides is high. The negative effects of sulfides on higher plants
include the following:

1. The direct toxicity of free sulfide as it comes in contact with plant roots.
2. The reduced availability of sulfur for plant growth because of its precipitation
with trace metals; and
3. The immobilization of zinc and copper by sulfide precipitation.

In wetland soils that contain high concentrations of ferrous iron (Fe$^{2+}$), sulfides
are reduced with iron to form insoluble ferrous sulfides (FeS), thus reducing the
toxicity of the free hydrogen sulfide. Ferrous sulfide gives the black color characteristic
of many anaerobic wetland soils. One of its common mineral forms is pyrite, FeS$_2$, the
form of sulfur commonly found in coal deposits.

The Carbon Cycle
The major processes of carbon transformation under aerobic and anaerobic conditions
are shown in Figure 5.12. Photosynthesis and aerobic respiration dominate the aerobic
processes, while anaerobic processes dominate the anaerobic processes.