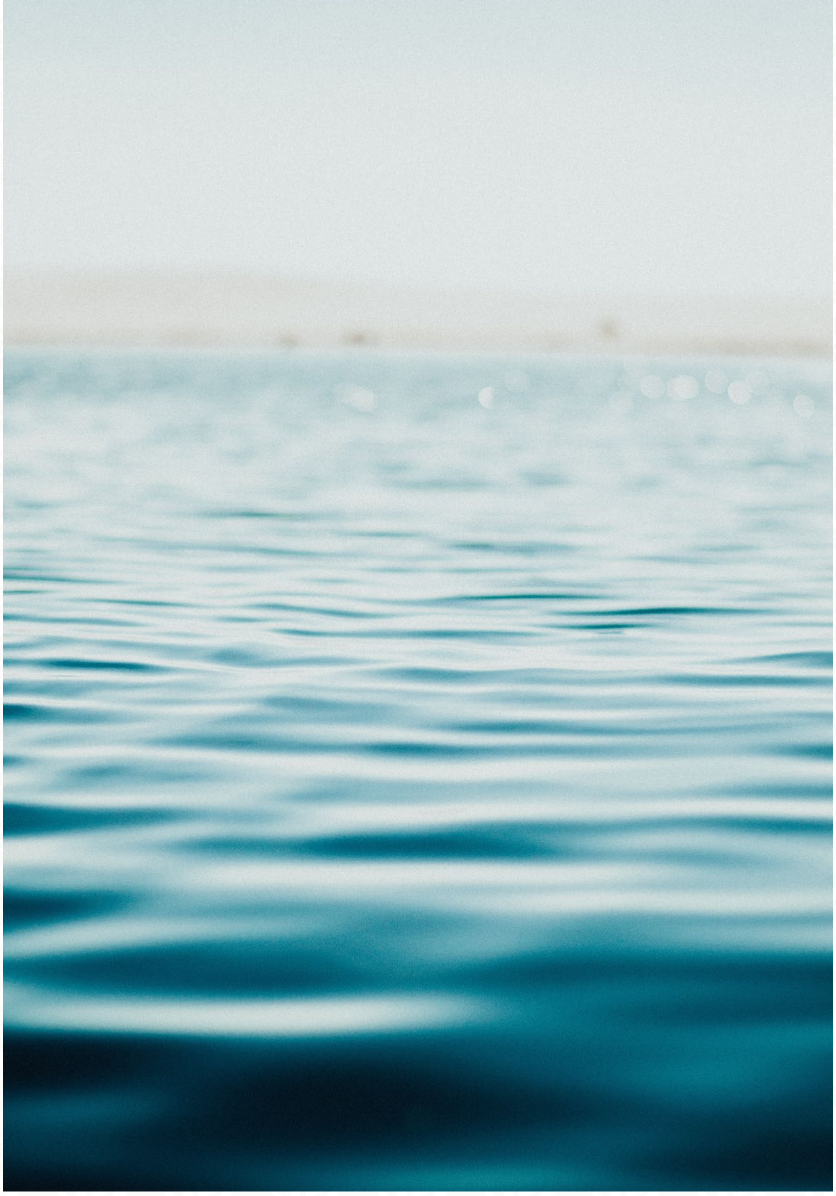


Nr: B2464
Mars 2023

En kunskapssammanställning: Avloppsvattenrening för att reducera kväve i kalla avloppsvatten med MBBR-processen

Kunskapssammanställning av Hallvard Ødegaard (SET) och Bjørn Rusten (Aquateam Cowi).
Svensk sammanfattning av Linus Karlsson (IVL)

I samarbete med: MittSverige Vatten och Avfall (MSVA), Miljö och Vatten i Örnsköldsvik AB (MIVA), Luleå miljöresurs AB (LUMIRE), Vatten och Avfallskompetens i Norr AB (Vakin), Purac AB och Scandinavian environmental technology AS (SET)

Författare: Kunskapssammanställning av Hallvard Ødegaard (SET) & Bjørn Rusten (Aquateam Cowi)
Svensk sammanfattning av Linus Karlsson (IVL)
Medel från: Stiftelsen Institutet för Vatten- och Luftvårdsforskning (SIVL), Baltic Sea Action Plan genom Nordiska Miljöfinansieringsbolaget (NEFCO) och Svenskt Vatten Utveckling
Fotograf: Alice Castor
Rapportnummer: B2464
ISBN: 978-91-7883-488-4
Upplaga: Finns endast som PDF-fil för egen utskrift

IVL Svenska Miljöinstitutet AB, Box 210 60, 100 31 Stockholm
Tel 010-788 65 00 // www.ivl.se

Rapporten har granskats och godkänts i enlighet med IVL:s ledningssystem

Förord

Flera reningsverk i norra Sverige står inför ombyggnationer eller flytt av reningsverken. Samtidigt kan krav på dessa reningsverk komma att skärpas med reduktionskrav på totalkväve och ammonium. Det finns idag få praktiska erfarenheter att rena bort kväve från kallt avloppsvatten och det saknas en sammanställning av de samlade erfarenheterna. Projektet Kvävening vid kalla vatten (Kall-N) syftar till att i teori och praktik undersöka just denna fråga närmare.

I det första arbetspaketet i Kall-N-projektet har det aktuella forskningsläget sammanställts tillsammans med erfarenheter från de fullskaleanläggningar som finns i framför allt Norge. Utöver denna sammanställning körs ett långtidsförsök i en pilot med MBBR-teknik i projektet för att ta fram dimensionerande parametrar för design av en kvävereningsprocess för kallt avloppsvatten. Pilotkörningarna kommer även visa hur driftstrategierna kan optimeras för att minimera förbrukningen av energi och insatsvaror samt huruvida uppvärmning av avloppsvatten med spillvärme kan vara en kostnadseffektiv lösning.

I denna rapport avrapporteras det första arbetspaketet, d v s kunskapssammanställningen som har tagits fram av prof. em. Hallvard Ødegaard, Scandinavian environmental technology AS (SET) och tekn. dr. Bjørn Rusten, Aquateam COWI. Den svenska sammanfattningen har skrivits av Linus Karlsson på IVL Svenska Miljöinstitutet (IVL) och projektet leds av tekn. dr. Andriy Malovanyy från IVL. Rapporten inleds med den svenska sammanfattning och därefter följer kunskapssammanställningen i sin helhet.

Utöver IVL så bidrar flera parter med finansiering och tid till projektet. Pilotanläggningen körs på Fillan reningsverk i regi av MittSverige Vatten och Avfall (MSVA). De andra VA-organisationerna som bidrar till projektet är Miljö och Vatten i Örnsköldsvik AB (MIVA), Luleå miljöresurs AB (LUMIRE) och Vatten och Avfallskompetens i Norr AB (Vakin). Utöver dessa deltar även Purac AB i projektet och extern finansiering har erhållits från Svenskt Vatten Utveckling, Stiftelsen Institutet för Vatten- och Luftvårdsforskning (SIVL) samt från Baltic Sea Action Plan genom Nordiska Miljöfinansieringsbolaget (NEFCO). Parallellt med projektet studeras även hur kväverening i kalla vatten påverkar utsläpp av lustgas till luften och utsläpp av hormoner och östrogenpåverkande ämnen till recipienten.

Projektgruppen hoppas att denna rapport ska bidra till att kunskapsläget ökar avseende kväverening i kalla avloppsvatten så att kostnadseffektiva och moderna anläggningar kan byggas på de reningsverk som i framtiden får kvävekrav.

Sammanfattning

Rapporten är uppdelade i fyra kapitel och inleder med en introduktion till MBBR-tekniken. Därefter följer en genomgång av hur MBBR-processen kan implementeras för avskiljning av kväve och fosfor i kallt avloppsvatten. I kapitel tre ges en sammanställning av genomförd forskning med studier från Norge, Kanada, USA, Italien och Sverige. Kapitel fyra djupdyker sedan i design och prestanda för fyra norska fullskaleanläggningar där MBBR-processen används för att reducera kvävehalten i kallt avloppsvatten. Dessa är Lillehammer avloppsreningsverk (ARV), Nordre Follo ARV, Nerdre Romerike ARV och Gardemoen ARV. I arbetspaket 1 har även anläggningar med MBBR-teknik och kalla förhållanden eftersökts i Sverige och Finland. I Sverige har Ängholmens reningsverk dessa förhållanden, men då anläggningen är dimensionerad för en mångdubbelt högre belastning än den faktiska under den kalla perioden, blir inte temperaturen begränsande för reningsverkets prestanda. I Finland togs Ruka ARV med MBBR-teknik i bruk 2016. Denna anläggning skulle vara väl lämpad för att utvärdera tekniken under kalla förhållanden, men då ingen extra provtagning har genomförts utöver de lagstadgade månatliga analyserna, är dataunderlaget för litet för en bedömning av reningsverkets prestanda under kalla månader.

Introduktion

Kallt avloppsvatten är vanligtvis en konsekvens av att smältvatten blandas med avloppsvattnet under töväder på våren. Under dessa perioder kan temperaturen i avloppsvattnet sjunka till under 5 grader under kortare perioder och ligga mellan 5 och 10 grader under många månader. Kalla temperaturer korrelerar därför vanligtvis även med ett utspätt avloppsvatten där tillgången på kolkälla för fördenitrifikation är begränsad. Inkommande avloppsvatten under snösmältningen kan även innehålla mycket löst syre vilket resulterar i nedbrytning av organiskt material i ledningsnätet. Detta minskar ytterligare tillgången på lättillgänglig kolkälla för fördenitrifikationen. Kallt avloppsvatten medför således designaspekter att ta hänsyn till utöver långsammare biologiska processer.

MBBR-processen i kalla vatten

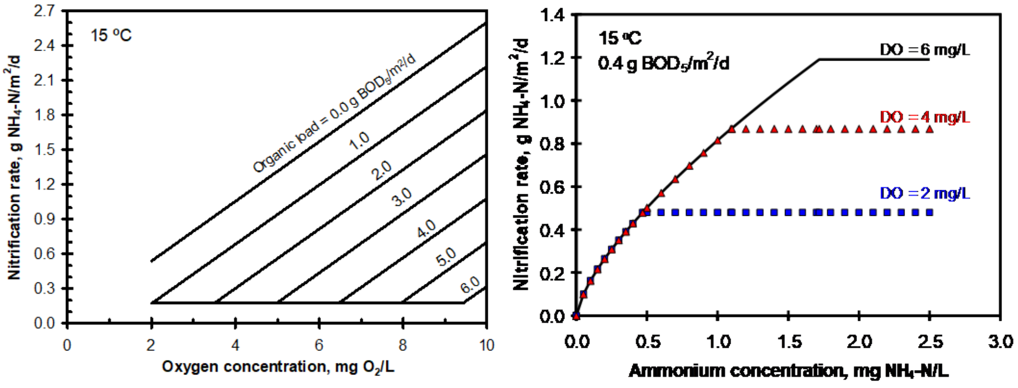
MBBR-processen kan designas på två principiellt olika sätt för att behandla kallt avloppsvatten beroende på lokala förhållanden. Om avloppsvattnet är koncentrerat används med fördel fördenitrifikation för att reducera behovet av extern kolkälla tillsammans med efterfällning. Om avloppsvattnet är utspätt och kallt, vilket under vintern och våren vanligtvis sammanfaller med snösmältning, återfinns en stor andel av det organiska materialet i partikulär form på grund av begränsad hydrolys i ledningsnätet. Syrehalterna i inkommande vatten är då också höga. För ett avloppsvatten med sådan karaktär så kan det vara mer lämpligt att använda förfällning tillsammans med efterdenitrifikation, eftersom tillgången på en lättillgänglig kolkälla i inkommande avloppsvatten är begränsad. Den lättillgängliga kol som ändå finns i inkommande avloppsvatten förbrukas av de höga syrehalterna i inkommande avloppsvatten i stället för av recirkulerad nitrat.

En kombination av för- och efterdenitrifikation kan också implementeras för att hantera förändringar i avloppsvattnets sammansättning och temperaturvariationer över året. Under snösmältningen förfälls vattnet och kväve avskiljs med efterdenitrifikation och när temperaturen ökar övergår reningsverket till fördenitrifikation kombinerat med efterfällning. Denna processlösning implementerades för första gången när Lillehammer byggde ut sitt reningsverk inför vinter-OS 1994 och var så pass framgångsrik att den har implementerats på ett flertal platser i Norge och övriga världen. Ett par av de biologiska zonerna byggs då variabla med både omrörare och luftning och nitratreturen kan regleras för att endast cirkulera den mängd nitrat som kan reduceras med det lättillgängliga kol som finns kvar efter försedimenteringen.

Kväverening vid kalla temperaturer ställer också höga krav på design och övervakning av syrehalterna i vattnet. När avloppsvattnet är utspätt och kallt, behöver nitratreturen begränsas och syrehalterna succesivt minska i de aeroba zonerna för att syre inte ska återföras till fördenitrifikationen eller resultera i en ökad konsumtion av extern kolkälla. En modern MBBR-anläggning utrustas därför med online-mätare för att optimera syretillförsel, nitratretur och dosering av extern kolkälla. En detaljerad designapproach för MBBR-processen gås igenom i denna rapport.

Forskningssammanställning

Studier på MBBR-processen påbörjades i Norge i slutet av 80-talet och visade hur betydande syrehalten i vattnet och den organiska belastningen är för nitrifikationshastigheten. Under denna period togs sambanden som presenteras i Figur 1 fram, som än idag utgör grunden för design av nitrifikationsprocessen i en MBBR-anläggning (Hem et al, 1994).



Figur 1. Nitrifikationsprocessens beroende av det organiska innehållet i avloppsvattnet och samspelet med mellan reaktionshastighet, syre- och ammoniumhalt i en MBBR-reaktor.

I de s k FAN-pilotförsöken som genomfördes på Nordre Follo ARV testades också hur MBBR-processen kan implementeras med kallt utspätt avloppsvatten (Rusten et al, 1995a; Rusten et al, 1995 b). Dessa försök visade att nitrifikationshastigheten är högre i ett förfällt vatten med låg partikelhalt och lite BOD jämfört med ett vatten som endast har genomgått försedimentering. Försöken låg till grund för den numera vedertagna regeln att designa zonen för BOD-reduktion före nitrifikationszonerna för en belastning på 5 g BOD_5/m^2,d. FAN-projektet bekräftade också tidigare slutsatser om nitrifikationshastigheten linjära beroende av syrehalten vilket också innebär att nitrifikationskapaciteten kan styras med syretillförseln och användning av höga syrehalter, som vanligtvis inte används i aktivslamprocessen, kan vara motiverat. Eftersom syrets löslighet ökar när temperaturen minskar samt att den endogena respiration (och följaktigt syrebehov) minskar vid sjunkande temperaturer, kan en högre syrehalt uppnås under kalla perioder med oförändrat luftflöde. Då blir temperatureffekten på nitrifikationen heller inte så stor som man kan tro.

Försöken visade att den observerade nitrifikationshastigheten var oberoende av temperaturen i intervallet 8 – 18 °C. I FAN-projektet undersöktes även hur utspätt avloppsvatten påverkar processen och försöken visade att en stor nitratretur kan minska fördenitrifikationen vid kalla temperaturer. Detta beror på att mer syre finns löst i nitratreturen och att anoxa förhållanden blir svåra att uppnå när den inkommande kol/kväve-kvoten är låg.

MBBR-tekniken fortsatte att utvecklas i Norge i och med att Lillehammer ARV byggdes om inför OS 1994. Under uppstarten av anläggningen under vintern 94/95 var temperaturen runt 6 – 6,5 °C. Under denna period var fördenitrifikationen begränsad och det mesta av denitrifikationen skedde med extern kolkälla (etanol) med en hastighet av 0,71 g NO_x-N/m^2,d. Nitrifikationshastigheten uppgick till 0,51 g NH_4-N/m^2,d med en syrehalt på 5 – 6,5 mg/l. Reduktionen av oorganiskt kväve låg under perioden på 82 % . I en annan studie på Frevar ARV ett par år senare kördes en MBBR-pilot under 2 års tid. Försöken visade att nitrifikationen berodde av temperaturen med en temperaturkoefficient på θ=1.07 i Arrhenius ekvation, se nedan, där k_{T1} är reaktionshastigheten vid temperaturen T_1 och k_{T2} är reaktionshastigheten vid en referenstemperatur T_2. Ett högre värde på temperaturkoefficienten, d v s theta i ekvationen, innebär ett större temperaturberoende.

$$k_{T_1} = k_{T_2} * \theta^{T_1 - T_2}$$

Om syrehalten hölls konstant uppgick dock koefficienten till θ=1.09 och motsvarande temperaturkoefficient för denitrifikationen uppgick till θ=1,05 i temperaturintervallet 5 – 17 °C. Men även denitrifikationsprocessen påverkas av syres ökade löslighet med minskande temperaturer vilket resulterar i höga inkommenade syrehalter samt höga syrehalter i nitratreturen när avloppsvattnet är kallt. Med höga syrehalter kommer en större andel av det organiska materialet i vattnet konsumeras med syre som elektronacceptor av de fakultativa anaeroba mikroorganismerna som normalt återfinns i fördenitrifikationen. Den verkliga temperaturkoefficienten i fördenitrifikationen uppgick i Frevarstudien till θ=1,10. I linje med tidigare experiment ökade också behovet av kolkälla när temperaturen sjönk, delvis eftersom höga syrehalter i inkommande avloppsvatten minskar möjligheten att använda BOD i avloppsvattnet för denitrifikation, men också eftersom en större mängd kol är partikulärt bundet då hydrolysprocessen är begränsad i ledningsnätet vid låga temperaturer. Den höga SS/BOD-kvoten vid låga temperaturer, samt behovet av extern kolkälla vid kalla förhållanden utöver det organiska materialet som finns i avloppsvattnet, innebär att slamproduktionen i biosteget blir större än när avloppsvattnet är varmare (Rusten et al, 2000).

Utöver forskning i Norge har studier även genomförts i andra länder. I Alaska uppgraderades Palmer ARV med syresatta dammar till en MBBR-process för en mer driftsäker process. Reningsverket designades för en lägsta temperatur på 5 °C och MBBR-processen klarade att reducera ammoniumhalten till under 1 mg/l under testperioden på 60 dagar (Figdore et al, 2019). Dock var temperaturen i genomsnitt 8,8 °C under detta test. På Johnstown ARV installerades en MBBR-anläggning för att förbättra nitrifikationen i ett dammsystem. Under vintern 04/05 var vattentemperaturen 4,2 °C och MBBR-processen reducerade ammoniumhalten med 40 %. Under en kort period sjönk temperaturen till 3 °C och pH ökade till 9. Nitrifikationsprocessen återhämtade sig kort efter att förhållandena återgick till normala (Wessman and Johnson, 2006). I Kanada har kväverening vid kalla temperaturer fått stort fokus då många reningsverk i landet består av dammsystem som inte kan reducera kvävehalten eller upprätthålla nitrifikationen under årets kalla månader. Studier har primärt genomförts i labb och visat att nitrifikationen är kraftigt beroende på temperaturen, även med höga syrehalter i vattnet. Studierna har också visat att nitrifierare acklimatiserar sig till låga temperaturer vilket resulterar i att nitrifikationshastigheten ökar med tiden (Delatolla et al, 2009; Delatolla et al, 2010). Vidare såg forskarna i Kanada att nitrifikation även sker vid mycket låga temperaturer, såsom 1°C, även om en nedre gräns för nitrifikationen på runt 2 – 4 °C identifierades eftersom reaktionshastigheten kraftigt minskar därunder. De kanadensiska studierna föreslog en temperaturkoefficient på θ=1,049 och θ=1,149 för temperaturintervallet 4 – 10 °C respektive vid 1°C (Hoang et al, 2014; Ahmed et al, 2019).

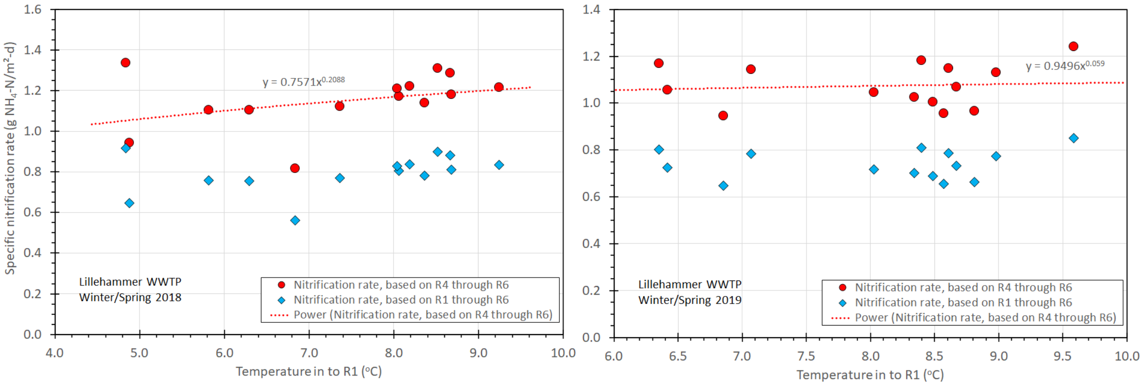
I Italien studerades MBBR-processen i full skala då ett reningsverk på en turistort i norra Italien uppgraderades. MBBR-reaktorerna placerades efter befintlig biologisk rening och två scenarier testades: ammoniumbegränsning och syrebegränsning. Vid ammoniumbegränsning kunde ingen korrelation mellan nitrifikationshastigheten och temperatur ses medan vid syrebegränsning sågs en korrelation med minskande hastigheter vid låga temperaturer, helt i linje med de norska studierna (Andreottola et al, 2000). I Sverige har en studie fokuserat på denitrifikation under kalla temperaturer och ett svagt samband identifierades då reaktionshastigheten vid 3 °C uppgick till 55 % av hastigheten vid 15 °C (Welander & Mattiasson, 2003).

Fullskaleanläggningar i Norge

Lillehammer avloppsreningsverk

Som tidigare nämnt uppgraderades Lillehammer ARV till en MBBR-anläggning inför vinter-OS 1994. Anläggningen designades flexibel där både kombinationen förfällning/efterdenitrifikation och fördenitrifikation/efterfällning är möjliga. Av 9 biologiska zoner kan 7 drivas aerobt. Reaktorerna är fyllda med K1-bärare (AnoxKaldnes) och är 5,5 m djupa.

Under 2018 var medeltemperatur i avloppsvattnet 10,8 °C och som lägst 3,9 °C och under året låg reduktiongraden av kväve på 80 % över reningsverket. Under den kallaste perioden ökade flödet från runt 10 000 m³/d till över 50 000 m³/d. Nitrifikationshastigheten i anläggningen har beräknats både med hela den aeroba volymen och med en mindre volym baserat på att föregående reaktorer används för BOD-reduktion. Generellt var nitrifikationshastigheten hög under 2018 då verket drevs med fördenitrifikation och påverkades bara marginellt av temperaturen, se vänster graf i Figur 2. Hastigheten för efterdenitrifikation minskade kraftigt med temperaturen under året men om detta berodde på en utspädning av avloppsvattnet, driftstrategin eller temperaturen är osäkert. Under 2019 var kvävereduktionen 75 %, medeltemperaturen 10 °C och den lägsta temperaturen 5,3 °C. Under den kallaste perioden ökade flödet till över 20 000 m³/dag. Nitrifikationshastigheten varierande heller inte med temperaturen under detta år och visas i den högra grafen i Figur 2. Värdena som presenteras Figur 2 är framtagna från veckoprover från en kallare period under respektive år.



Figur 2. Nitrifikationshastighet och dess korrelation med temperatur under åren 2018 (tv) och 2019 (th) på Lillehammer ARV. Datapunkterna är framtagna med veckomedelvärden från den kallare perioden under respektive år.

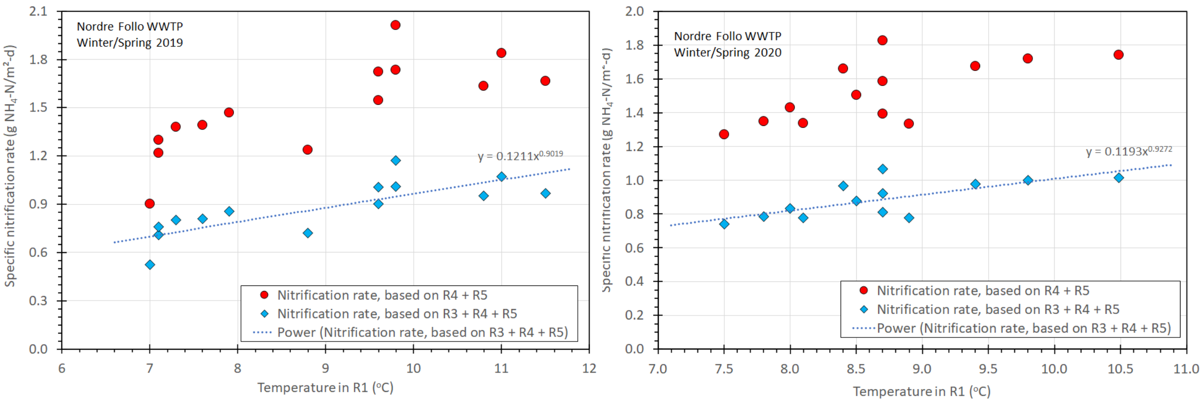
När avloppsvattnet är varmare blir processen begränsad av substrat vilket gör att nitrifikationshastigheten inte ökar med en högre temperatur. Under 2019 var hastigheten för efterdenitrifikation oberoende av temperaturen. Jämfört med 2018 var inkommande avloppsvatten dock mindre utspätt under 2019 vilket kan ha orsakat skillnaden i temperaturberoende jämfört med föregående år. Sammanfattningsvis visar resultaten från Lillehammer ARV att den observerade nitrifikationshastighet kan vara stabil eller minska något vid kallare temperaturer.

Gällande temperatureffekten på denitrifikationen är det svårt att dra slutsatser eftersom reningsverkets driftstrategi under tillfället inte är känd och utspädningen under snösmältning verkar ha haft stor påverkan på hastigheten. Även för nitrifikationshastigheten är det svårt att bedöma vilka parametrar och driftstrategier som har haft betydelse för det relativt svaga beroendet av temperaturen. Framför allt saknas det syrehalterna för en djupare förståelse av processen.

Nordre Follo avloppsreningsverk

Nordre Follo ARV uppgraderas 1997 till en MBBR-anläggning med försedimentering och fördenitrifikation. Bioreaktorerna är 5,5 m djupa och fyllda med K1-bärare (AnoxKaldnes). Under slutet på 2018 togs även en deammonifikationsreaktor (AnitaMox, Veolia) i drift för rejektvattenbehandling. På reningsverket har denitrifikationsprocessen historiskt varit begränsande vilket har kompenserats genom att metanol sporadiskt tillsätts i inloppet till biologin.

För att beräkna reaktionshastigheterna i biologin har kvävebelastningen på biologin antagits vara samma som inkommande kväve till reningsverket. Under 2019 reducerades inkommande kväve med 80 % på reningsverket trots att belastningen översteg designvärdet. Motsvarande siffra för 2020 var 79 %. Även på detta reningsverk är nitrifikationsprocessen substratbegränsad förutom under årets kalla månader. Under vinterhalvåret 2019 och 2020 låg medeltemperaturen på runt 9 - 10 °C och ner mot ca 7 – 7,5 °C under höga flöden 2019. Under 2020 var den lägsta temperaturen något högre. Beräknad nitrifikationshastighet och dess korrelation med temperaturen visas i Figur 3 för 2019 (tv) och 2020 (th). Beräknade hastigheter baseras på den totala aeroba volymen, samt på att en zon används för BOD-reduktion.



Figur 3. Beräknad nitrifikationshastighet på Nordre Follo ARV under 2019 januari till maj (tv) och 2020 januari och april (th).

Nitrifikationen beror här på temperaturen och korrelationen med Arrhenius ekvation visar på en temperaturkoefficient på θ=1.10 under 2019 i temperaturintervallet 7,0 – 11,5 °C och θ=1,11 under 2020 i temperaturintervallet 7,5 – 10,5 °C. Denitrifikationens beroende är svårt att bedöma eftersom befintlig data inte möjliggör att separera kväve avskilt med fördenitrifikation med kväve avskilt med efterdenitrifikation. Vidare har metanol sporadiskt tillsatts till inloppet, vilket ytterligare komplicerar en bedömning av denitrifikationshastigheten.

Nedre Romerike avloppsreningsverk

Nedre Romerike avloppsreningsverk (NRA) byggdes 2003 med försedimentering och kombinerad för- och efterdenitrifikation följt av efterfällning och avskiljning av bildade flockar med sedimentation. För efterdenitrifikation används vanligtvis metanol, förutom under vissa perioder då denna kolkälla kompletteras med beslagtagen etanol. MBBR-anläggningen är fylld med bärare av typen K1 (AnoxKaldnes) och under 2018 ersattes försedimenteringen med så kallade bandfilter (Salsnes).

I detta projekt har data från kalla perioder under åren 2013, 2014, 2016 och 2018 analyserats närmare för att bedöma hur den biologiska reningsprocessen fungerar med kallt avloppsvatten. Medeltemperaturen under dessa perioder låg på 8,6 °C och den lägsta temperaturen på mellan 5,2 – 7,4 °C. Nitrifikationshastigheten varierade mellan 0,31 – 0,41 g NH_4-N/m^2,d beräknat på hela den aeroba volymen då reningsverket inte har någon separat zon för BOD-reduktion. Reaktionshastigheten i fördenitrifikationen beräknades till mellan 0,18 – 0,22 g NO_x-N/m^2,d och i efterdenitrifikationen till 1,0 – 1,1 g NO_x-N/m^2,d. Temperaturkoefficienten för båda processerna uppskattades till θ=1,07 – 1.11 i temperaturintervallet 5 – 10 °C.

Kvävereduktionen har generellt varit god på reningsverket under kalla temperaturer förutom då även inkommande flöden är höga och avloppsvattnet utspätt. Då minskar reduktionen kraftigt till runt 50 % över reningsverket vilket indikerar att effekten av ett utspätt avloppsvatten är mer betydande för kväveavskiljningen än temperaturen.

Gardemoen avloppsreningsverk

Gardemoen ARV färdigställdes 1998 samtidigt som Oslos nya flygplats öppnade nästgårds. Reningsverket behandlar kommunalt avloppsvatten och flygplatsens avloppsvatten, samt uppsamlad avisningsvätska sedan 2004. Den mest utspädda avisningsvätskan genomgår en biologisk förbehandling i en separat MBBR-anläggning innan den når reningsverket. Den mer koncentrerade strömmen används på reningsverket som extern kolkälla. Eftersom insamlad avisningsvätska är mycket kall har en värmeväxlare installerats för att värma denna ström med renat avloppsvatten. Huvudprocessen på Gardemoen ARV består av rensgaller och sand- och fettfång följt av försedimentering. Det biologiska steget drivs med kombinerad för- och efterdenitrifikation i MBBR. Efter biosteget tillsätts fällningskemikalier och partiklar avskiljs i en flotationsanläggning. Före avloppsvattnet når recipienten desinficeras det med UV-ljus under badsäsongen.

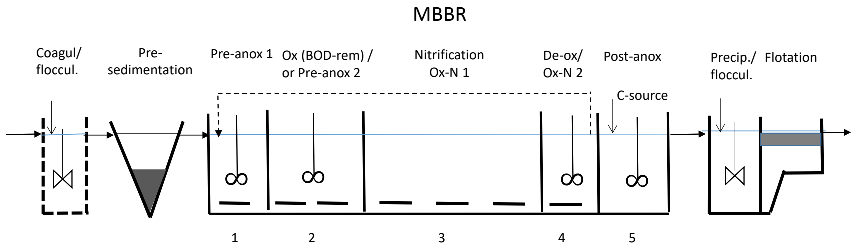
För bedömning av reningsverkets prestanda finns ett stort antal dygnsprov på ingående och utgående vatten att tillgå från 2019 och 2020. Under dessa år reducerades inkommande kvävehalt med 85 %, trots att kvävebelastningen översteg designvärdet. Data visar att denitrifikationshastigheten beror av ett flertal faktorer och ett temperaturberoende därför ibland är svårt att identifiera. Data indikerar dock att reaktionshastigheten i fördenitrifikationen under perioden låg på i genomsnitt 0,88 g NO_x-N/m^2,d i temperaturintervallet 8,0 – 16,5 °C och minskade till 0,68 g NO_x-N/m^2,d i temperaturintervallet 5,4 - 8 °C. Dessa hastigheter är något högre än förväntat. Hastigheterna i efterdenitrifikationen kunde inte korreleras med temperaturen och låg i genomsnitt på 1,2 g NO_x-N/m^2,d, vilket kan jämföras med 1,4 g NO_x-N/m^2,d som är den högsta förväntade reaktionshastigheten med den använda kolkällan. Nitrifikationshastigheten varierade svagt med temperaturen under perioden och en temperaturkoefficient på θ=1,015 i temperaturintervallet 6 – 16 °C har beräknats för de stunder när processen inte har varit begränsad av tillgången på ammonium. Även i detta fall verkar det som att syrets ökade löslighet i kallt avloppsvatten, lägre endogen respiration samt högre syrehalter döljer reaktionens verkliga temperaturberoende. Eftersom reningsverkets driftstrategi inte är dokumenterad kan det svaga temperaturberoendet också bero på t ex ökad luftning i de aeroba biostegen vid kalla temperaturer.

Design för kallt avloppsvatten

I de norska designriktlinjerna har MBBR-processen beskrivits sedan 2009. Dessa designriktlinjer uppdaterades så sent som 2020[1]. Tillgänglig forskning och den datasammanställning som har gjorts i detta projekt visar att de norska riktlinjerna lämpar väl sig för att nå en tillräcklig kvävereduktion även i anläggningar som tar emot kallt avloppsvatten. Det innebär att en anläggning där utgående totalkvävehalter i genomsnitt ska understiga 10 mg/l eller kvävereduktionen ska överstiga 70 % kan designas för 10 °C även om temperaturen i avloppsvattnet understiger 10 °C under långa perioder och stundvis sjunker så lågt som till 5 °C. Dock bör designen för 10 °C kontrolleras mot ett scenario med kallt utspätt avloppsvatten.

Vidare så har genomgången av tidigare studier och databearbetningen från fullskaleanläggningar att en flexibel anläggning med kombinerad för- och efterdenitrifikation är att föredra där kallt utspätt avloppsvatten förekommer. I en sådan anläggning kan både det organiska materialet som finns i avloppsvattnet utnyttjas och driftsäkerheter uppnås vid behov genom att tillsätta extern kolkälla. I Figur 4 visas ett konceptuellt exempel på en sådan anläggning.

Kunskapssammanställningen har också visat på kunskapsluckor vad gäller hur driftstrategier påverkar de biologiska reaktionshastigheternas beroende av temperaturen. Sammanställningen visar att temperaturberoendet skiljer sig mycket åt mellan reningsverk och mellan labb-studier. Detta förklaras delvis med förändringar i vattenmatrisen vad gäller näringskoncentrationer, syrehalter och det organiska materialets tillgänglighet för de biologiska processerna när avloppsvattnet är kallt och utspätt. Dock påverkar också reningsverkens driftstrategier, såsom syretillförsel, dosering av extern kolkälla och nitratretur. För att ta fram lämpliga driftstrategier och underlag för att bedöma investerings- och driftkostnader för svenska förhållanden, kommer dessa parametrar studeras noga i pilotstudien i detta projekt.



Figur 4. Konceptuell design av en flexibel MBBR-anläggning för att behandla kallt avloppsvatten.

---

[1] Norskt Vann (2020) *Veiledning for dimensionering av avlopsrenseanlegg*. Rapport No 256.

Wastewater treatment for N-removal in MBBR-based plants receiving cold wastewater

A knowledge compilation

scandinavian environmental technology a.s.

Prof. em. dr.ing. Hallvard Ødegaard

Preface

This report is the result of an assignment that SET AS has carried out on behalf of IVL Swedish Environmental Research Institute under the project: “Nitrogen in cold wastewater – Long term pilot experiments with MBBR”.

The purpose of the project is to increase knowledge about nitrogen removal in cold wastewater by the use of the MBBR technology in order to achieve cost-effective treatment at low temperatures. The project consists of 6 work packages out of which this report belongs to work package AP1 Knowledge compilation.

The report is written by prof. em. Hallvard Ødegaard, SET AS and dr.ing. Bjørn Rusten, Aquateam COWI, as a sub-concultant to SET AS. Both have long experience with MBBR. Prof.em. Ødegaaard is the inventor of the MBBR and its processes and he has worked closely together with dr.ing Bjørn Rusten ever since the first pilot experiments were carried out in Norway. There are more experiences with the use of MBBR for nitrogen removal in cold wastewater in Norway than in any other country in the world.

The report is divided in four chapters. After the introduction (Chapter 1); the moving bed biofilm reactor (MBBR) and MBBR-based processes for N- and P-removal in cold wastewater are presented in chapter 2, focusing on system design and reactor partitioning. In chapter 3 a review of international publications, dealing with studies of N-removal in MBBRs (mostly lab- and pilot) at low temperatures, is given including studies from Norway, Canada, USA, Italy and Sweden. In chapter 4 nitroen removal in full-scale MBBR plants at low temperatures in Norway during recent years are presented and discussed. These are Lillehammer WWTP, Nordre Follo WWTP, Nedre Romerike WWTP and Gardermoen WWTP.

The authors are grateful to the plant owners that have let the data from their plants to the disposal for the authors, which has made it possible for us to present the most comprehensive experience study on this subject ever.

Content

1 Introduction

Cold wastewater is normally a consequence of intrusion of snow-melt water into the wastewater pipes during snow-melting in the spring. Occasionally the temperature of the incoming water to treatment plants in Scandinavia may fall under 5°C, and quite commonly plants experience temperatures between 5 °C and 10 °C for a longer or shorter period. When the effluent standard is based on yearly average (for instance 10 mg tot N/l in effluent or 70 % reduction based on yearly average), it is quite common to design treatment plants in Scandinavia for 10 °C. This is, for instance, the case for the Norwegian Design Guidelines (Norsk Vann, 2020).

This situation results in several items that has to be taken into account in design and operation:

1. The biological processes on which N-removal depends (nitrification and denitrification) proceed at a low rate, the lower the temperature is. Luckily, there is some rate (even though low) all the way down to 1 °C, but we seldom has to take lower temperatures than 5 °C into account. The temperature dependency of microbial processes are often described by the Arrhenius temperature correction coefficient, θ (see chapter 2.3).
2. The wastewater is normally diluted by the snow-melt water. On one hand this makes it easier to meet a given effluent standard concentration value, but on the other hand it will be more difficult to meet a percentage removal standard
3. Pre-denitrification is dependent on availability of easily biodegradable organic matter in the raw wastewater. In a diluted wastewater the amount of easily, biodegradable organic matter is lower and hence the capacity to denitrify is lower. And also the concentration of easily biodegradable organic matter is lower and hence so is the driving force for denitrification. Therefore the denitrification rate is lower in a snow-melt diluted water.
4. In a cold snow-melt situation the wastewater may be aerobic, i.e. have some oxygen in it. This will in the first place result in an aerobic biofilm in the network that consumes some easily biodegradable organic matter and in the second place, consume easily biodegradable organic matter in the pre-denitrification tank, that otherwise would have been used for denitrification – with an inferior denitrification as the result.

Hence there are several challenges in planning, designing and operating nitrogen removal plants for cold wastewater situations.

In this report we shall in chapter 2 discuss favorable reactor designs of the moving bed biofilm reactor (MBBR) that have been used in the wastewater characteristic situation described. In chapter 3 we shall summarize experiences from studies published in literature, concerning N-removal in MBBR plants and chapter 4 we shall present and discuss data compiled from existing MBBR-plants for N-removal in Norway in recent years. In chapter 5 our findings are summarized.

The design criteria that are most used, are those proposed by Norsk Vann (Norsk Vann, 2020). The chapter herein that deals with MBBR, is attached to this report (in Norwegian)

2 The moving bed biofilm reactor (MBBR) and MBBR-based processes for N- and P-removal in cold wastewater

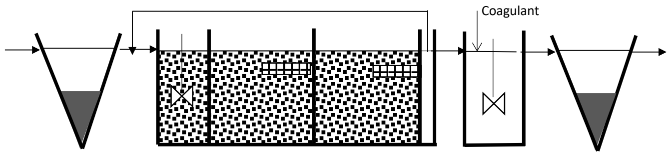
2.1 The MBBR and MBBR-based systems for N- and P-removal

There are basically two extreme designs for the design of MBBR-plants for nutrient (N and P) removal, as shown in Figure 2.1a and 2.1b:

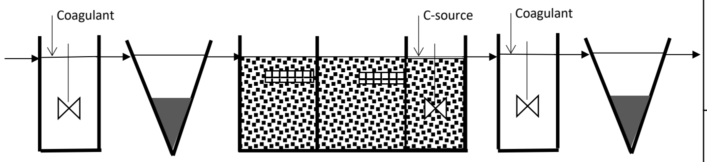
1. Pre-denitrification with post-precpitation (Figure 2.1a), that is most suitable when the water is concentrated and has a high content of easily biodegradable, soluble organic matter, i.e. that the $\mathrm{COD}_{\mathrm{biodegradable}}$/N-ratio in the incoming water is high
2. Pre-precipitation with post-denitrification (Figure 2.1b), that is most suitable when the water is diluted and has a high content of organic matter in particulate form and hence that the $\mathrm{COD}_{\mathrm{biodegradable}}$/N-ratio in the incoming water is low

In situations where cold water is a challenge in parts of the year (primarily during snow-melting in spring), it is typical that the water has a relatively high content of organic matter on particulate form. This is partly caused by the fact that the water is oxygen-rich resulting in aerobic degradation in the network that transforms soluble, easily biodegradable organic matter into particulate organic matter (bacteria cells). This is evident by the fact that many such plants achieve a very high removal of particulate organic matter by chemical coagulation (pre-coagulation) alone (> 70%).

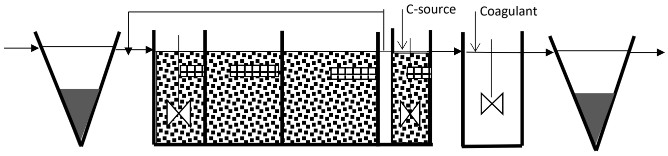
a. Pre-denitrification with post-precipitation



b. Pre-precipitation with post-denitrification



c. Combuned pre- and post-denitrification



Advantages and disadvantages
Advantages • No need for external carbon source
Disadvantages • Limitation to what can be achieved in terms of removal efficiency (max ca 70 %) • Dependent on a relatively high and stable C/N in • Dependent on recycle, i.e. high Q through the bioreactor
Avantages • Not dependent on C/N-ratio in in-coming water • No limitation with respect to treatment efficiency (> 90 %) of both N and P • Low Q through the bioreactor (less area for sieves)
Disadantages • Need for external carbon source • Pre-precipitation must be controlled in order to prevent P-limitation in bioreactor
Advantages • Little dependent on C/N-ratio in • No limitation with respect to treatment efficiency (> 90 %) of both N and P
Disadvantages • Need for external carbon source • Is dependent of recycle, i.e. high Q through bioreactor

Figur 2.1. Process solutions for removal of N and P in MBBR-plants.

In most situations where cold wastewater is a challenge, each of the two extremes may be more suitable at different periods of the year, pre-preciptation and post-denitrification during winter/spring months and pre-denitrification and post-precipitation during the summer/autumn months. Hence a third alternative, the combined pre- and post-denitrification MBBR (Figure 2.1c) is being favored in plants with large variation in wastewater composition over the year.

The first plant with this reactor design was built for the wastewater treatment plant in Lillehammer, Norway, in connection with the winter Olympic Games in 1994. The experiences with this process solution was so favorable that all the major nitrogen plants based on MBBR in Norway now uses the combined pre- and post-denitrification process, and it is now also being favored on plants in other countries (Sweden, Finland, Uruguay, China (including Hong Kong) and Australia as well.

A typical build-up of a MBBR based on pre- and post-denitrification is shown in Figure 2.2. As demonstrated some of the reactors are so-called “swing”-reactors, i.e. they have both a system of aerators as well as mixers, meaning that they may be operated both aerobically and anoxically. Together with a smart control system, this makes the combined pre-and post-denitrifcaion MBBR system very flexible and ideally suited for situations with large variations in temperature and flow.

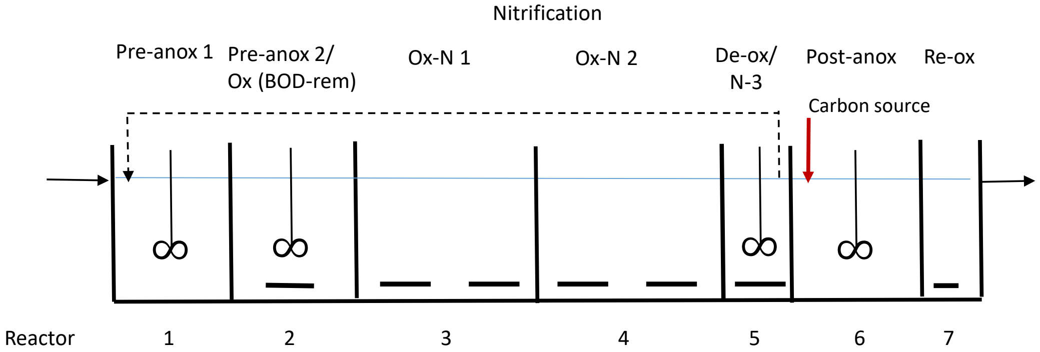


Figure 2.2. Typical build-up of a MBBR based on pre- and post-denitrification.

The first reactor is normally a mixed, anoxic pre-denitrification reactor that receives nitrate containing water from the end of the nitrification zone (reactor 5 in Figure 2.2). The recycle flow is made variable since it it does not help to recycle more than the amount of easily biodegradrable organic matter in the raw water can accommodate. The recycled water contains oxygen (coming from reactor 5) and the more one recycles, the more oxygen will be used to consume the precious, easily biodegradable organic matter intended for nitrate removal. Hence with the kind of water we are designing for in the “Cold N”-plants, the recycle will normally not be set higher than $2*Q_{in}$ ($r = Q_r/Q_{in} = 2$). Theoretcally, this can lead to a potential removal of nitrogen in the pre-denitrification equal to approximately $r/(r+1)*100\%$, i.e. $2/3*100 = 67\ \%$. In cold, diluted wastewater (as during snow melt in spring) the availability of easily biodegradable, organic matter can be so low that the recycle may be set lower, for instance at $\leq 1Q_{in}$, equivalent to a theroretical removal of < 50 %.

The second reactor stage is normally a swing reactor that may be operated with mixing alone, and become a second anoxic reactor, or with aeration alone as a BOD-removal reactor. During summer time (or dry weather flow), the availability of easily, biodegradable organic matter in the raw water (and hence the pre-denitrification capacity) is higher. Inflow is also lower so a higher recycling ratio may be set – all in all resulting in a higher N-removal in the pre-denitrification stage. And the benefit for this, is a lower consumption of external carbon source for the subsequent post-denitrification in order to meet the overall N-removal goal.

The third reactor is aerobic and used for BOD-removal and possibly partly for nitrification while the fourth reactor is definitely for nitrification. In summer time, when temperatures and nitrification rates are higher and pre-denitrification is maximized by using reactor 2 for pre-denitrification, reactor 3 and 4 together will have sufficient aerobic capacity to both remove enough BOD to start nitrification and to nitrify to the extent targeted. In the winter and especially in the snow-melt situation in spring, nitrification rate is lower and BOD-removal takes primarily part in reactor 2 that is now aerobic.

Reactor 5 is another “swing” reactor. It is, however, normally mixed and not aerated even though it will be aerobic because of the relatively high DO-concentration coming into the reactor from reactor 4. We want the oxygen concentration out of the reactor as low as possible, in order to minimize the consumption of organic matter, that otherwise would be utilized for nitrate removal either in the pre- or post-denitrification reactors. There is, however, not much substrate left for oxygen consumption, but correctly designed there will be some ammonium left that will consume oxygen at a relatively low rate. At this point the nitrification process is ammonium limited, however, so low DO does not limit the ammonium degradation.

Reactor 6 is a post denitrification reactor where external carbon source is added. The denitrification rate here is much higher than in the pre-denitrification stage, so this reactor is relatively small, even though it may be the most important one in order to achieve a certain treatment goal. If the requirement is very low nitrate concentration in the effluent, reactor 6 may be divided in two steps, but this is not common when tot N-removal of around 70-80 % is aimed for.

Reactor 7 is normally an aerated, small reactor with the task to remove any excess organic matter from the added carbon source. If the control of carbon source is good or if carbon is made the limiting factor for denitrification, this reactor may be omitted. That is also the case if the MBBR is followed by coagulation/dissolved air flotation.

Improved process control is often established by using on-line sensors. The nitrification rate is strongly dependent on DO and by regulating the air supply from the DO-signals, nitrification e.g.in reactor 4 may be controlled. A sensor measuring nitrate in reactor 6 may be used to control the addition of external carbon source. The experiences from Norwegian combined pre- and post-denitrification plants (see chapter 4) have shown that the post-denitrification step has been crucial for safeguarding a good treatment result. All in all, the wastewater situation that may be encountered in cold and wet weather situations, calls for the use of a post-denitrification step, because this makes one independent upon the wastewater characteristics of the incoming water. If using a swing reactor also in the first stage of the MBBR (see Figure 2), one may also facilitate the extreme version of a pure post-denitrification system that may be used in extreme situations.

2.2 Design criteria of the pre- and post-denitrification MBBR

The dimensioning of the pre- and post-denitrification MBBR is carried out based on design rates for each of the reactor stages; pre-denitrification, BOD-removal, nitrification, de-oxygenation and post-denitrification. The necessary reactor volume is determined based on the design rates ($g/m^2_{biofilmarea}\cdot d$), the effective, specific area (m^2/m^3) of the carrier that is to be used, and the filling fraction (%) of the carrier.

The design procedure is as follows:

1. The amount of nitrogen that is to be nitrified and denitrified is determined based on the influent and effluent concentrations, the assimilation during biodegradation (based on BOD-removed) and recycle rate.
2. The amount of easily biodegradable organic matter available for pre-denitrification is determined based on the incoming soluble BOD-concentration and the soluble BOD-concentration that is created by hydrolysis. The amount easily biodegradable organic matter is the sum of the two.
3. The NO_x-N load on the pre-denitrification reactor is the sum of the amount of recycled nitrate and the converted oxygen-load (1 g O_2 is equal to 0.35 g NO_3-$N_{equivalents}$).

4. The volume of the pre-denitrification reactor is determined based on the design rate for pre-denitrification.
5. The residual BOD-load on the BOD-removal reactor is determined as the incoming load minus that amount that is used for the NO_x-N removal.
6. The volume of BOD-removal reactor is determined based on the design rate for BOD-removal.
7. The volume of the nitrification reactor(s) is determined based on the design nitrification rate that is set depending on the pre-treatment.
8. The volume of the de-oxygenation reactor is determined based on the design nitrification rate for de-oxygenation.
9. The amount of post-denitrification needed is determined as the difference between the amount of nitrogen to be denitrified and the amount denitrified in the denitrification reactor.
10. The volume of the post-denitrification reactor(s) is determined based on the design post-denitrification rate that is set.

In addition to determining the reactor volumes, the air needed has to be determined as well as the biomass separation reactor.

The design criteria that are most used, are those proposed by Norsk Vann (Norsk Vann, 2020). The chapter herein that deals with MBBR, is attached to this report (in Norwegian)

2.3 The Arrhenius temperature correction coefficient

The Arrhenius temperature correction coefficient, θ, is commonly used for calculating microbial process rates at one temperature as compared to another, using the equation below:

$$k_2 = k_1 \cdot \theta^{(T_2 - T_1)}$$

where k_1 and k_2 are the process removal rates at temperatures T_1 and T_2 (°C), respectively

There are actually very few studies of temperature effect at low temperatures on nitrification and denitrification in MBBRs. Most references on temperature influence are from plants with temperature above 15 °C.

The θ-value for MBBR-processes is often set at around 1,09 for nitrification processes and 1.07 for denitrification and BOD-removal processes (with 10 °C as reference temperature) as it is done in the Norwegian design guideline (Norsk Vann, 2020). When analyzing the literature more closely, these are quite conservative values. Hem et al (1994) found, for instance, a value of 1.044 (in the 8-18 °C range) and Salvetti et al (2006) 1.058 for an oxygen limited attached growth nitrification.

Although the equation above can be used to model the temperature effects on the nitrification rates, it does not take into account another important factor; the exposure time. Delatolla et al. (2009) proposed a temperature and time dependent equation to model the attached growth nitrification rates as a function of elapsed time at 4 °C:

$$\theta = 3.81 \times 10^{-2} \ln(t) + 9.83 \times 10^{-1}$$

where t is the exposure time in days to a specific temperature.

To achieve efficient operation of nitrifying MBBR systems, it is important to evaluate the effects of cold-shocks on nitrification kinetics, not only to account for periods of snowmelt, but also the projected increase in frequency of extreme weather events and cold air outbreaks.

The influence of temperature on denitrification in MBBRs are even more seldom studied, but one may assume that the effect is the same as for heterotrophs that remove BOD.

3 Review of publications concerning studies of N-removal in MBBRs (mostly lab- and pilot) at low temperatures

3.1 Experiences from pilot-plants in Norway

In addition to the experiences from full-scale plants in the later years that are described in chapter 4, four important pilot- and full-scale studies in Norway from earlier times shall be mentioned here (Hem et al, 1994; Rusten et al 1995a; Rusten et al 1995b and c and Rusten et al, 2000).

3.1.1 Nitrification studies at NTH (now NTNU)

The studies of the MBBR process started already around 1988 with several master students studying nitrification and de-nitrification. The first study that was published, however, where the experiments that Lars Hem did for his PhD, fulfilled in 1991 (Hem et al, 1994). Hem did his experiments partly on a constructed wastewater and partly on real settled secondary effluent but he demonstrated that the relative effects on the different factors influencing the nitrification rate found for the constructed wastewater were also valid for the secondary treated effluent.

Hem demonstrated how important DO concentration and organic load are for the nitrification rate in MBBRs and he demonstrated that the ammonia concentration was only limiting for the nitrification rate at O_2/NH_4-N ratios above 10 mg O_2/mg NH_4-N, which in practice means at ammonia concentration as low as 1-2 mg NH_4-N (depending on DO – see Figure 3.2).

Based on Hems observations, the much-used plots in Figure 3.1 and 3.2 were constructed; that in many ways sum up Hem's thesis work and forms the ground for the design of nitrification in MBBRs as we use it today.

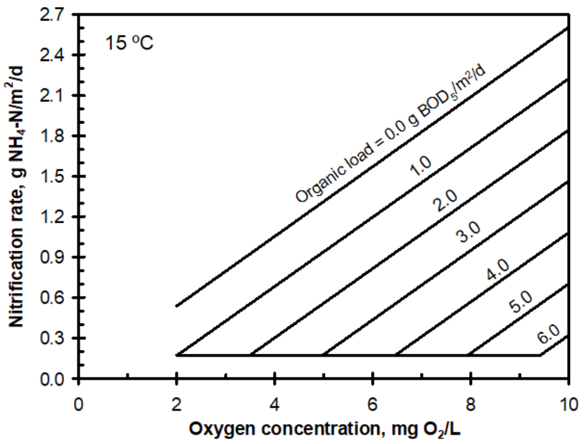


Figure 3.1. Nitrification removal rate taking both organic load and DO concentration into consideration at 15 °C (Temp. corr. ϴ = 1.09).

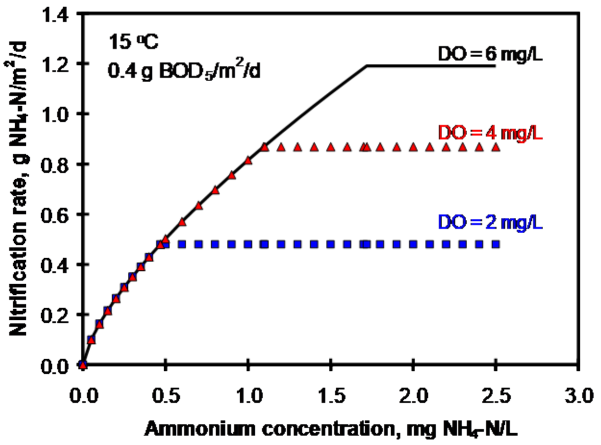


Figure 3.2. Nitrification removal rate versus ammonium concentration at different DO-levels at low organic load (Temp. corr. ϴ = 1.09).

Hem did most of his experiments at room temperature, but did some short-term experiments at temperatures down to 8 °C. Above ca. 12 °C he found a temperature coefficient θ = 1.044 (in the equation $k_2 = k_1 \cdot \theta^{(T_2 - T_1)}$) and in the 8 – 12 °C range he found θ = 1.09.

3.1.2 The FAN pilot plant experiments.

The FAN pilot plant experiments were carried out at Nordre Follo WWTP as part of the "Removal of Nitrogen program" in Norway (FAN-program, 1988-1992) (Ødegaard, 1992). The results of these studies are extensively reported in the journal Water Environment Research (Rusten et al, 1995b and 1995c). Here we shall concentrate on the issues that are especially important in treatment of cold and diluted water.

Pre-denitrification with post-precipitation and post-denitrification with pre-precipitation were studied. These experiments were the first ones on nitrogen removal in MBBRs in the world and the information collected forms the basis for our understanding of the MBBR and its design.

The pilot-plant consisted of two parallel trains, one for each of the two MBBR systems; a. pre-denitrification with post-precipitation and b. post-denitrification with pre-precipitation (see Figure 3.3)

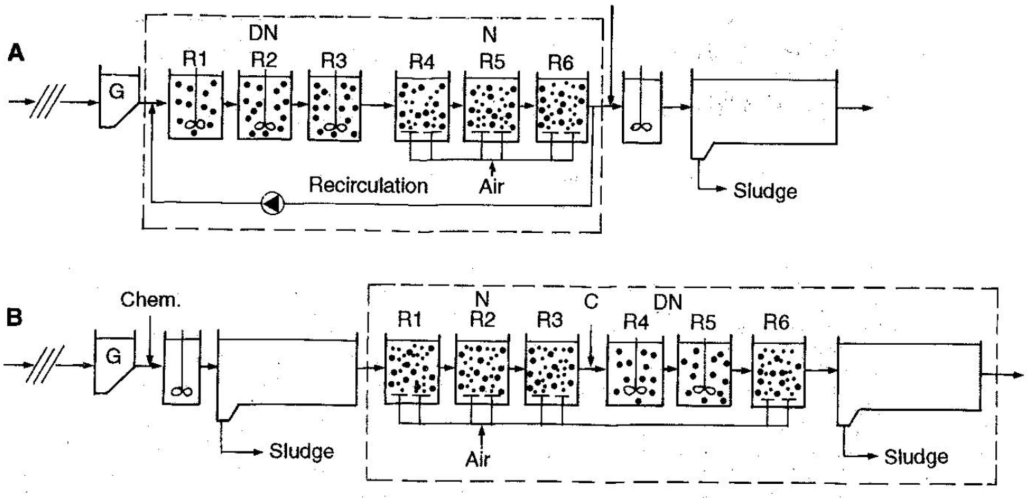


Figure 3.3. The pilot plants lay-out in the Nordre Follo WWTP experiments.

The pilot reactors had an active specific biofilm are of approximately 310 m^2/m^3 and were operated at water temperatures of 7 to 18 °C. The raw municipal wastewater had a very low concentration of easily biodegradable soluble COD (BSCOD) and had a high DO concentration.

3.1.2.1 Nitrification.

Figure 3.4 show the overall nitrification rates versus ammonium loads for the: a) pre-denitrification system based on the biofilm surface in all the aerobic reactors and b) the post-denitrification system based on the biofilm surface area of the three aerobic reactors. The figure shows that the nitrification rate, calculated based on the total aerobic volume of the two systems, seemed to be on the same level. In the post-denitrification/pre-precipitation system, however, there is easily biodegradable organic matter left to be removed aerobically in the aerated stages of the bioreactor, while this has mostly been removed anoxically before the aerobic reactors in the pre-denitrification/post-precipitation system.

When rates are calculated on the purely nitrifying reactors, the maximum nitrification rate in the post-denitrification/pre-precipitation system (1.6 g NH_4-N/m^2d – see Figure 3.6) was demonstrated to be higher than in the pre-denitrification/post-precipitation system (1.0 g NH_4-N/m^2d). This is the reason for higher design nitrification rate in systems with advanced particle removal (pre-precipitation) in the Norwegian design guidelines.

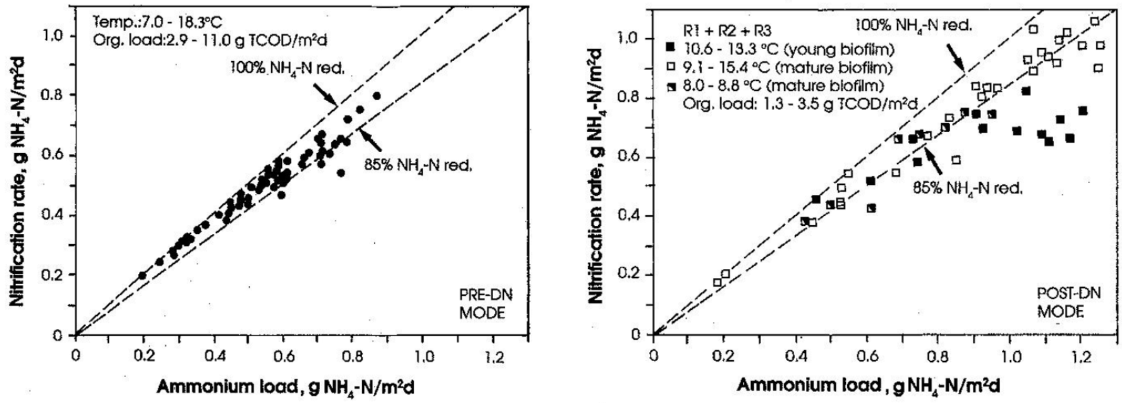


Figure 3.4. Overall nitrification rates versus ammonium loads for the a) pre-denitrification system based on the biofilm surface in all the aerobic reactors and b) the post-denitrification system based on the biofilm surface area of the three first aerobic reactors.

The philosophy for design is therefore to design for a BOD-removal reactor that removes so much of the biodegradable organic matter that nitrification is not limited by organic matter, before the nitrification reactors are designed. As Figure 3.5 shows, this was demonstrated to be around 4.5 g BSCOD/m^2d (around 10 g COD/m^2d) that has been transformed to 5 g BOD_5/m^2d in the Norwegian design guidelines.

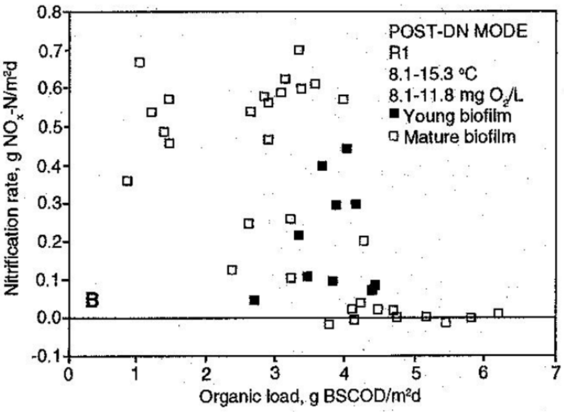


Figure 3.5. Nitrification rate versus organic loading.

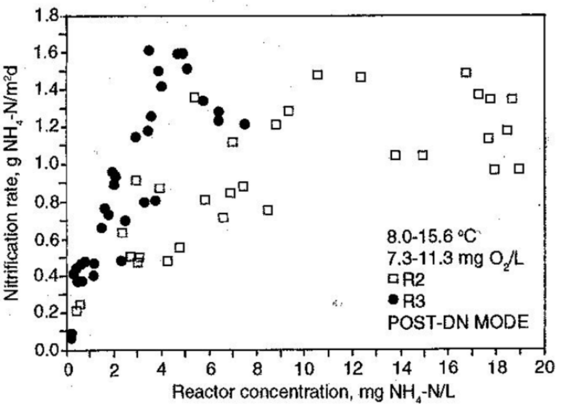


Figure 3.6. Nitrification rate versus NH_4-N concentrations in the 2. and 3. reactor in the post-denitrification system.

Figure 3.6 shows nitrification rate in the post-denitrification system at high DO, demonstrating that the nitrification rate was limited by ammonium only at quite low ammonium concentrations (< 3 mg NH_4-N/l) as also demonstrated earlier by Hem (Hem et al, 1994). This is because the nitrification rate at higher concentrations is not limited by ammonium but oxygen.

Oxygen is actually the most important factor that determines the nitrification rate in MBBR's in practice – once the biodegradable matter has been removed upstream (aerobically or anoxically). This is well demonstrated in Figure 3.7 based on short-term tests during the Nordre Follo WWTP experiments.

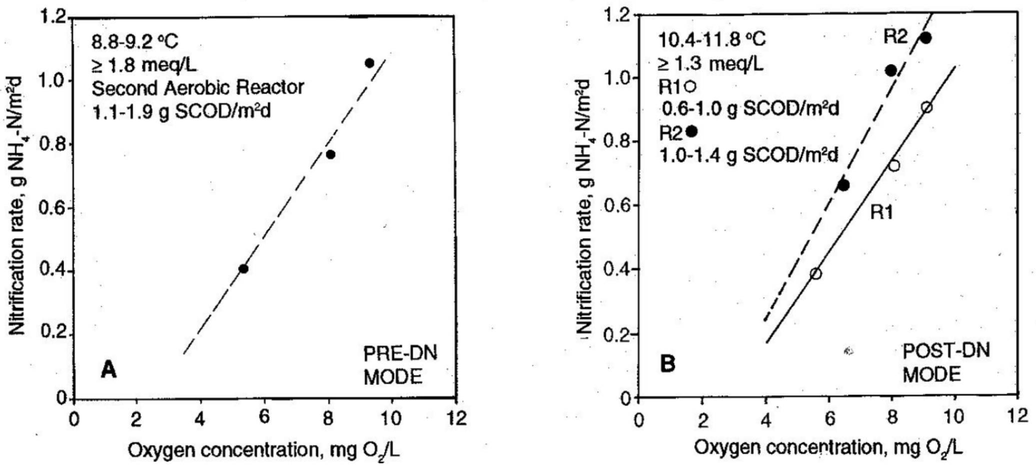


Figure 3.7. Nitrification rate versus oxygen concentration, based on short-term tests with a) the second aerobic reactor in the pre-denitrification system and b) the first two reactors in the post-denitrification system. The removal of organic matter during the test is indicated in the figures.

A linear relationship between rate and DO is clearly demonstrated, all the way up to at least 10 mg O_2/l and the lines cross the x-axis at around 2.5 mg O_2/l. The good thing here is that nitrification rate in the MBBR may be regulated by the DO (hence air supply) in the reactor – the bad one being that you need at least a DO of 2.5 mg/l in the reactor. When relating to cold weather, however, one benefit comes for free. When the temperature goes down, the solubility of DO goes up, and so does the nitrification rate. Therefore, in practice, the apparent influence of temperature on nitrification is quite low – as many researchers have demonstrated.

In the Nordre Follo WWTP experiments, the apparent temperature effect on nitrification was indeed insignificant in the range of 7 to 18 °C. Accounting for the differences in DO concentrations, a true temperature coefficient of θ = 1.09 was found.

In Figure 3.8 is shown the nitrification rate versus ammonium concentrations at two different runs in pre-denitrification and post-denitrification mode, at temperatures below 10°C and at similar and high DO-concentrations. Dotted lines show nitrification rates based on the reaction rate constants and the NH_4-N concentrations for transition from oxygen to ammonia rate limitation

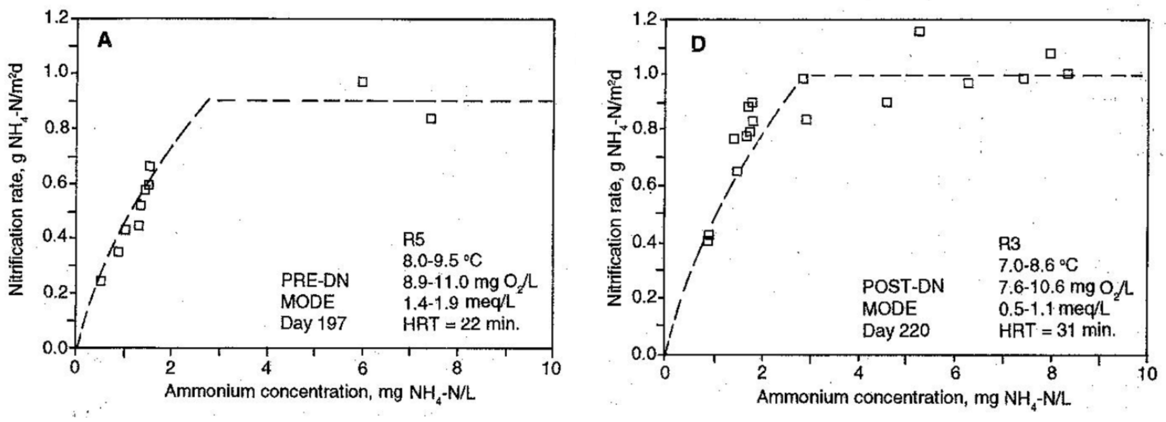


Figure 3.8. Nitrification rate versus ammonium concentrations at two different runs in pre-denitrification and post-denitrification mode and at temperatures below 10°C. Dotted lines show nitrification rates based on the reaction rate constants and the NH_4-N concentrations for transition from oxygen to ammonia rate limitation.

It is demonstrated that the maximum nitrification rate is higher in the post-denitrification system and that the results fits well with the model proposed:

$$r_N = k \cdot (S_N)^n$$

r_N = reaction rate constant, g NH_4-N/m^2d
S_N = concentration of NH_4-N in the reactor, mg/l
k = reaction rate constant
n = reaction order constant

The reaction rate constant at ammonia limitation was dependent on the mode of operation and was between 0.44 and 0.71 assuming the typical reaction order constant for biofilms of 0.7.

This model formed the basis for design of nitrification in MBBRs. The n-value was recommended to be set at 0.7 and the k-value:

k = 0.40 no primary sedimentation or pre-denitrification
k = 0.47 with primary sedimentation or pre-denitrification
k = 0.53 with primary sedimentation and pre-denitrification
k = 0.58 with chemical precipitation

S_n is the lower of the NH_4-N concentrations found when using the actual NH_4-N concentration in the reactor, or the NH_4-N concentration at the point of transition where the rate limiting substrate change from ammonium to dissolved oxygen (DO). S_n at the point of transition can be found by the following equation:

$$S_{n,\text{ trans}} = \frac{\text{((Bulk liquid DO) – (DO depletion across heterotrophic part of biofilm*))}}{\text{(Autotrophic biofilm DO to } NH_4 \text{ -N ratio at transition**)}}$$

* In practice the DO depletion across the heterotrophic part of the biofilm will vary depending on the growth conditions for the biofilm and the soluble BOD concentration in the reactor. This depletion can vary from 0.5-2.0 mg/l, but an average of 0.5 mg/l can be used for MBBR reactors with low (~ 2 mg/l) soluble BOD concentrations.
** This ratio (mg DO/l to mg NH_4-N/l) will vary according to literature, but 3.2 was recommended

So the formula for the NH_4-N concentration at the point of transition, $S_{n\text{ trans}}$, can be simplified to:

$$S_{n,\text{ trans}} = \frac{\text{(Bulk liquid DO - 0.5)}}{3.2}$$

where bulk liquid DO and S_n trans are concentrations in mg O_2/l and mg NH_4-N/l, respectively.

This corresponds well to a maximum nitrification rate at 10°C of 1.01 g NH_4-N/m^2d in the pre-denitrification system and 1.24 g NH_4-N/m^2d in the post-denitrification system at a high DO in the Nordre Follo pilot experiments.

Alkalinity may limit nitrification in biofilm processes. In general it has been recommended that a residual alkalinity of 1.5 mekv/l is necessary in order to avoid alkalinity limitation (with corresponding pH-fall inside the biofilm). In the Nordre Follo pilot experiments it was found, however, that considerably lower alkalinity (0.7-0.8 mekv/l) was acceptable and it was speculated that this might be caused by the extremely thin biofilm in nitrifying MBBRs.

In pre-precipitation systems too much phosphorus may be removed ahead of the MBBR causing P-limitation in the reactor. Based on the Nordre Follo WWTP experiments, it was recommended that a residual PO_4-P concentration of 0.3 mg P/l should be maintained.

3.1.2.2 Denitrification.

It was difficult to establish good pre-denitrification because of the low concentrations of easily biodegradable organic matter and high DO in the influent as well as in the recycled water. The overall pre-denitrification rate was always lower than 0.4 g NO_x-N/m^{2}d whereas the maximum denitrification rate of the first pre-anoxic reactor was close to 0.7 g NO_x-N/m^{2}d (see Figure 3.9). The scatter demonstrates how important the content of biodegradable, soluble COD (BSCOD) in the influent was for the obtained rate. The highest maximum denitrification rates required BSCOD > 24 mg BSCOD/l.

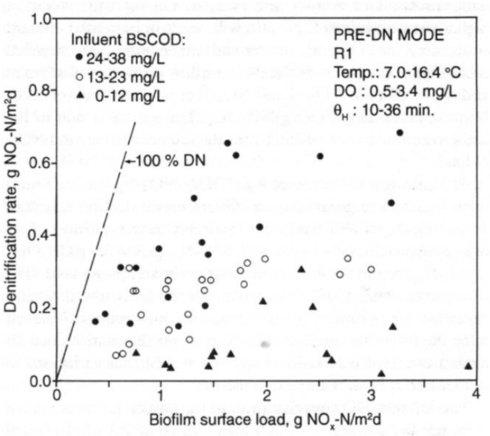


Figure 3.9. Denitrification rates for the first anoxic reactor (R1) in the pre-denitrification process at various BSCODs in influent.

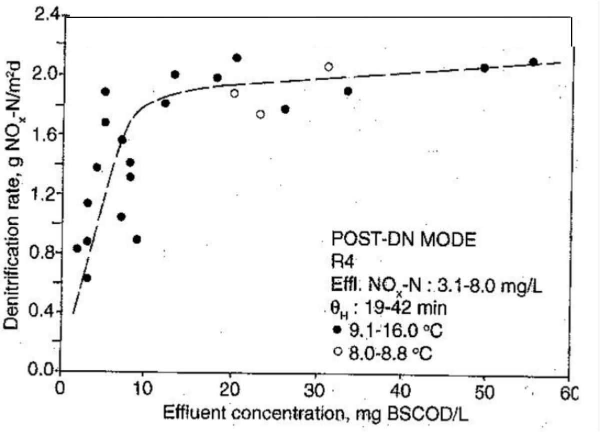


Figure 3.10. Denitrification rates vs effluent BSCOD concentrations for the first anoxic reactor (R4) on the post-denitrification process (at effluent NO_x-N concentrations > 3 mg/l).

Figure 3.10 shows the influence of BSCOD in the post-denitrification process. Already at BSCOD concentrations above 10 mg BSCOD/l in the effluent, the denitrification rate was close to maximum at a much higher level than in the pre-denitrification case, around 2.0 g NO_x-N/m^{2}d – even at temperatures lower than 10 °C. A C/N-ratio > 3 g COD/g NO_x-N was found to be needed in order to reach good denitrification with around 4 COD/g NO_x-N as the optimal.

Figure 3.11 demonstrates very well what will happen to the total N removal in a pre-denitrification process on diluted wastewater.

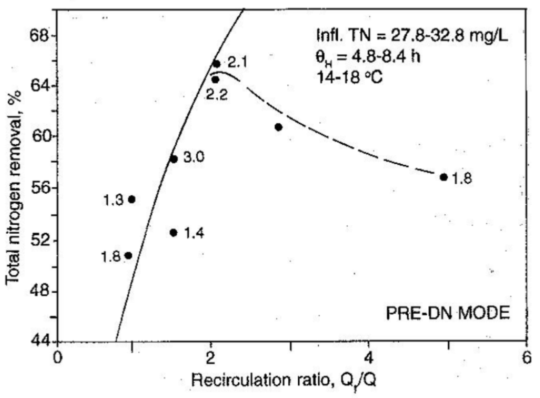


Figure 3.11. Total N removal efficiency versus recycling ratio for the pre-denitrification process at an anoxic zone equal to 50 % of the total biofilm area. Figures indicate C/N-ratio.

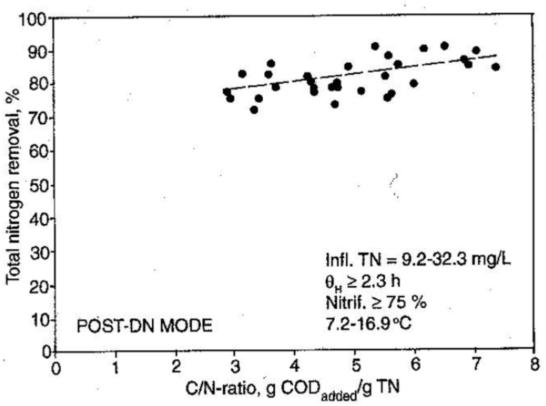


Figure 3.12. Total N removal efficiency versus C:N-ratio (g COD$_{added}$/g Tot N) in pre-precipitated water in the post-denitrification process.

Removal increases with increasing recycling ratio (because more NO_3-N is available for pre-denitrification), but at a certain recycle ratio (in this case Q_R/Q_{in} ~ 2) the removal starts to drop. This is caused by the fact that the amount of oxygen that is recycled, exceeds the relative amount of NO_3-N that can be removed in the fight for the easily, biodegradable organic matter available.

The line in the figure follows the equation that is based on a mass balance over the reactor (not considering assimilation) and can be used for rough estimates:

$R = r/(r+1)*100\%$

R = Total N removal that can be expected in pre-denitrification
r = recycle ratio = $Q_{recycle}/Q_{in}$

In the post-denitrification process one is independent of the raw water BSCOD and a high removal could be secured as long as sufficient sodium acetate as carbon source was added (> 3 g COD_{added} /g Tot $N_{influent}$) as demonstrated in Figure 3.12.

Figure 3.13 demonstrates well the influence of the C/N-ratio of effluent Tot N concentration (mg/l) and Tot N removal (%) in dilute wastewater, i.e. low temperature, low inlet concentration and short hydraulic residence time. Concentration below 7 mg tot N/l (at influent of 12.9 mg tot N/l) was obtained at a C/N-ratio less than 3 g COD_{added} /g Tot $N_{influent}$ at less than 2 hrs total HRT in the MBBRs in post-denitrification mode. An effluent concentration of only 3.3 mg tot N/l was reached at a C/N-ratio of 4.3 g COD_{added} /g Tot $N_{influent}$.

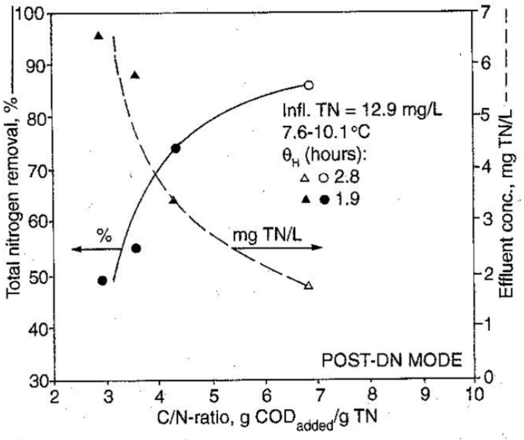


Figure 3.13. Total N removal and effluent concentrations versus C/N-ratio for the post-denitrification process treating dilute waste-water at low temperatures and short HRT.

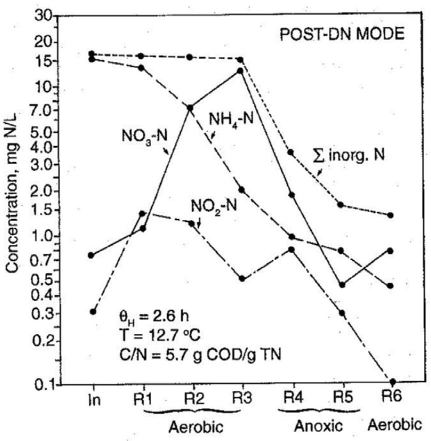


Figure 3.14. Example of inorganic nitrogen profiles through the pilot plant in the post-denitrification mode. Concentrations are given in log-scale.

Figure 3.14 demonstrates how the different inorganic components changed throughout the reactor in the pre-precipitation/post-denitrification process. Note that the total HRT of the MBBR was as low as 2.6 hrs in this example.

The specific sludge production (kg TS/kg COD) – determined over an 8 month period - was significantly lower in the MBBR of the post-denitrification process (pre-precipitated sludge not included). It was 0.59 kg TS/kg $COD_{applied}$ (0.73 kg TS/kg $COD_{removed}$) in the pre-denitrification process and 0.25 kg TS/kg $COD_{applied}$ (0.34 kg TS/kg $COD_{removed}$) in the post-denitrification process (COD as sodium acetate included).

3.1.3 The Lillehammer WWTP start-up tests

Lillehammer wastewater treatment plant was upgraded for the 1994 Winter Olympics. It was until then a traditional chemical secondary precipitation plant with pre-sedimentation, coagulation, flocculation and post-sedimentation. The upgrade consisted of building a MBBR for nitrogen removal between the pre-sedimentation tanks and the coagulation/flocculation tanks. A MBBR was chosen because of lower space requirements and lower price than the activated sludge alternative, and it became the first combined pre- and post-denitrification plant in the world. Because there were no experiences with that sort of set-up, it was built with a lot of flexibility with respect to reactor stages and the possibility to operate them with mixing, aeration or both (swing reactor) – se Figure 3.15

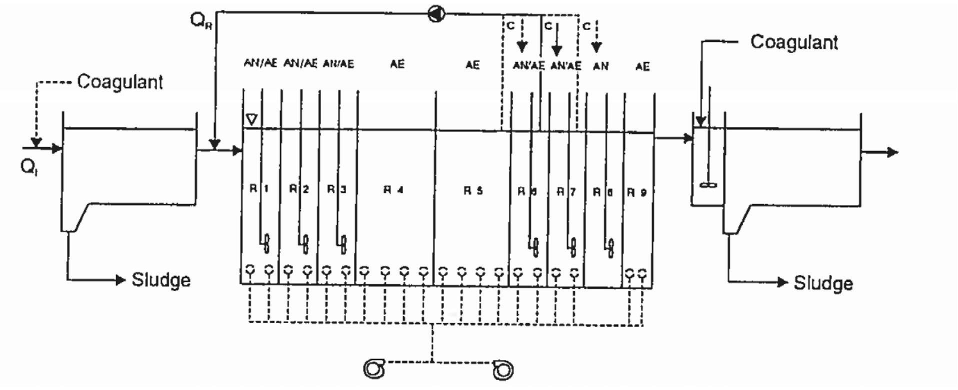


Figure 3.15. Reactor lay-out of the Lillehammer WWTP.

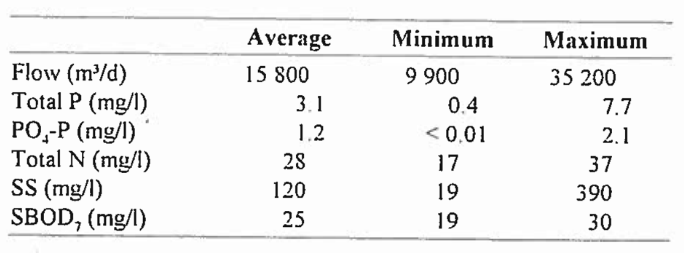
The MBBR has 9 stages altogether, R1-R3 are all swing reactors and may operate both anoxically and aerobically, R4 and R5 are aerated (for nitrification) while R6-R7 are swing reactors (for nitrification/de-oxidation) or post denitrification, R8 is mixed and not aerated (for post-denitrification) and finally R9 is for post-oxygenation. The plant has 2 trains and uses K1 carriers with a filling fraction of 65 % in all stages from the start-up.

The plant was equipped with NH_4-N and NO_3-N sensors to regulate aeration (DO and hence nitrification rate) and to control dosing of carbon source (hence control denitrification rate).

During the start-up period from January 1 to May 31, 1995 (i.e. a winter period), the plant was followed up very closely and below the experiences from this period shall be reported (see also Rusten et al, 1995a).

Flow rates and influent concentrations, based on 24h flow proportional samples, are shown in Table 3.1. All influent concentrations were very low during wet weather conditions, while concentrations of soluble phosphorus (PO_4-P) and soluble organic matter ($SBOD_7$) were low all the time.

Table 3.1. Flow rates and influent concentrations for the Lillehammer WWTP, Jan. 1 to May 31, 1995.



	Average	Minimum	Maximum
Flow (m^3/d)	15 800	9 900	35 200
Total P (mg/l)	3.1	0.4	7.7
PO_4-P (mg/l)	1.2	< 0.01	2.1
Total N (mg/l)	28	17	37
SS (mg/l)	120	19	390
$SBOD_7$ (mg/l)	25	19	30

The influent wastewater temperature was well under 10 °C most of the time during this period and for longer periods under 7 °C.

The MBBR of the first train was started up during winter conditions with primary treatment water in influent. The MBBR had been loaded with carriers (K1) since November 1994 and aeration had started December 1. The plant started to nitrify already after about two weeks and complete nitrification was achieved at the beginning of January 1995. The second MBBR train was seeded with nitrifies water from train one and started on March 5, 1995, and good nitrification was observed after two weeks in this train as well. Denitrification, with ethanol as the cheapest carbon source, was initiated on March 22 for train 1 and April 22 for train 2.

Results from operation with combined pre- and post-denitrification of primary treated wastewater are shown in Figure 3.16 and Table 3.2. Reactors 1 and 2 were anoxic reactors 3, 4 and 5 were aerated, reactor 6 was mixed by propeller. It was not aerated – but aerobic in order to reduce DO and nitrified wastewater was recycled from reactor 6 to reactor 1. Ethanol was added as external carbon source to reactor 7 and reactors 7 and 8 were anoxic while reactor 9 was aerated.

Data were collected from several days of 24h flow proportional sampling, beginning only 2 weeks after recycling and addition of external carbon source was initiated. Figure 3.16 shows an example of the profile of nitrogen and SCOD through MBBR train 1, with a recycling ratio (Q_{rec}/Q_{in}) of 1.0 at 6.5 ^{o}C.

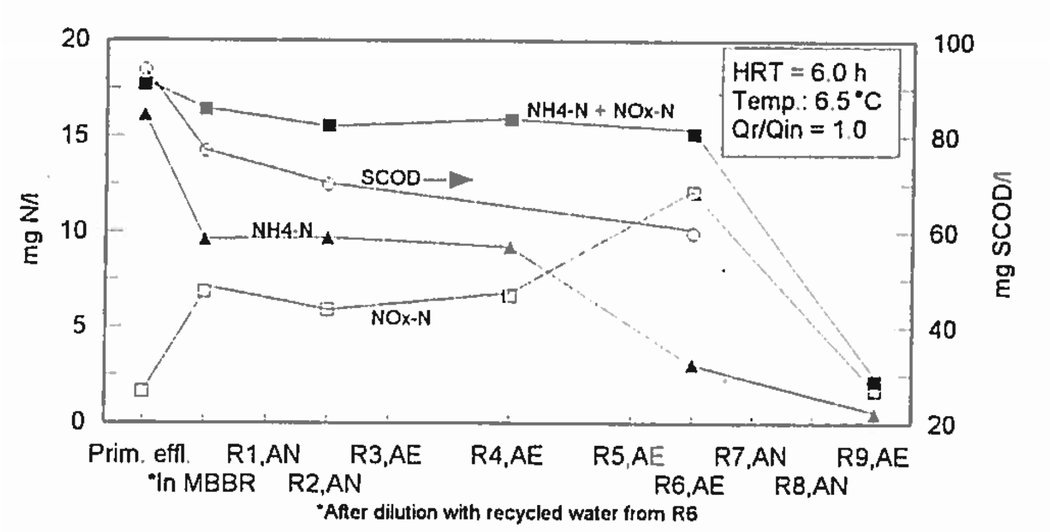


Figure 3.16. Example of nitrogen and SCOD profiles through MBBR train 1 at Lillehammer WWTP treating primary effluent in the combined pre- and post-denitrification mode on April 10, 1995.

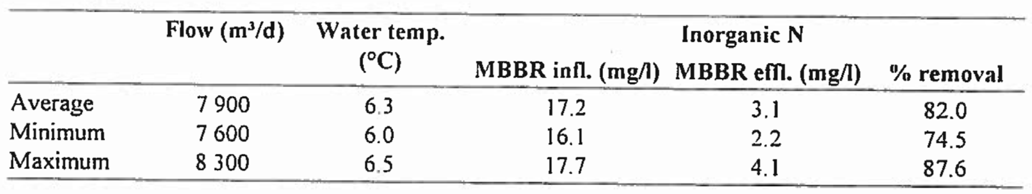
The very low temperature, high influent DO-levels and low amount of biodegradable, organic matter in the wastewater, resulted in low pre-denitrification activity. A consumption of 7.3 g SCOD/g NO_x-$N_{removed}$ was determined in the pre-denitrification stage.

Nitrification took place mainly in reactors 5 and 6 with a specific nitrification rate of 0.51 g NH_4-N/m^{2}d at a DO-level of 5.0-6.5 mg/l. This is as expected from the nitrification design model proposed (Rusten 1995b) – see under Nordre Follo experiments - and used in the Norwegian design guidelines.

The majority of the denitrification took place in reactors 7 and 8, with an average post-denitrification rate of 0.71 g NO_x-N/m^{2}d, slightly lower than recommended for design in the Norwegian design guidelines.

The overall removal of inorganic N during this start-up period with temperatures in the range of 6-6.5 ^{o}C is shown in Table 3.2.

Table 3.2. Removal of inorganic N in MBBR train 1 at Lillehammer WWTP, April 1995. Combined pre- and post-denitrification of primary effluent.



	Flow (m^3/d)	Water temp. (°C)	Inorganic N		
			MBBR infl. (mg/l)	MBBR effl. (mg/l)	% removal
Average	7 900	6.3	17.2	3.1	82.0
Minimum	7 600	6.0	16.1	2.2	74.5
Maximum	8 300	6.5	17.7	4.1	87.6

The average removal for inorganic N was 82 % for all the samples at these low temperatures (6-6.5 ^{o}C). The removal of Tot N would naturally be slightly higher since organic N in influent would be higher than in effluent. Of the 82 %, only 15-17 % was removed in the pre-denitrification stage because of the cold, snow-melt diluted wastewater.

3.1.4 The post-denitrification pilot test

Figure 3.17 shows post-denitrification rates in a pilot test carried out for the Gardermoen WWTP. Three different external carbon sources were tested (Rusten et al, 1996). The test was carried out at a situation where the nitrate concentration was not rate limiting. Use of methanol and mono propylene glycol (MPG) gave similar denitrification rates. Use of ethanol doubled the denitrification rates. All three external carbon sources showed a temperature coefficient of $\theta = 1.07$. Necessary C/N-ratios were also temperature dependent which can be explained by higher specific sludge production (lower decay) at lower temperatures. The necessary C/N-ratio increased with an average of 45 % when the temperature was reduced from 15 to 5 ^{o}C.

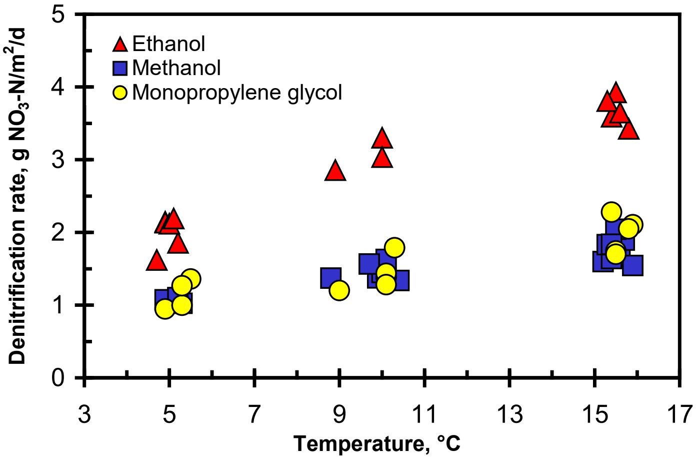


Figure 3.17. Denitrification rate versus temperature with various external carbon sources (Rusten et al, 1996).

For 95 % removal of nitrate, ethanol had the lowest carbon demand, with a necessary C/N-ratio of 3.9 g $COD_{added} \cdot$ g $NO_{3\text{-}Nequivalents}^{-1}$ at 10 ^{o}C. MPG had the highest necessary C/N-ratio at 5.6 g $COD_{added} \cdot$ g $NO_{3\text{-}Nequivalents}^{-1}$ at 10 ^{o}C (Rusten et al, 1996).

3.1.5 The FREVAR pilot plant experiments.

The pilot-plant study at FREVAR WWTP (Rusten et al. 2000) was carried out over 2 years. The pilot-plant layout was as shown in Figure 3.18. K1-carriers were used in all stages of the pilot plant at a filling fraction of 66 %, i.e. with a specific area of 330 m^2/m^3. The pilot-plant was operated in pre-nitrification mode only.

The study was carried out for both a summer and a winter situation – with wastewater characteristics that are quite similar to what can be expected for the “Cold N”-plants in Northern Sweden. A simplified flow sheet for the MBBR pilot plant used at FREVAR in Figure 3.18.

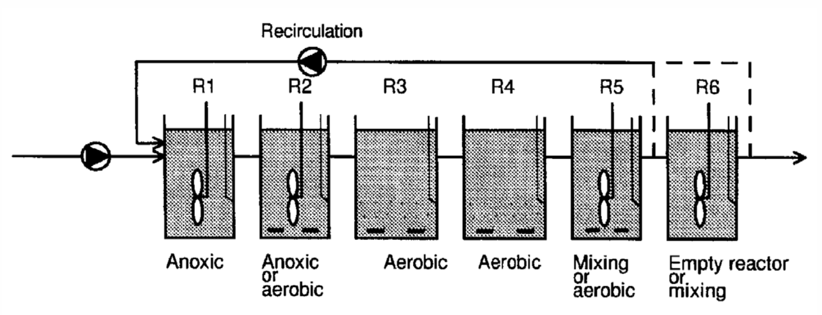


Figure 3.18. Simplified flow sheet for the MBBR pilot plant used at FREVAR.

In Table 3 examples of typical characteristics of primary treated effluent fed to the pilot plant are shown for a summer situation and a winter situation, respectively.

Table 3.3. Example of typical primary effluent concentrations fed to the pilot plant during a summer and a winter situation, respectively.

Parameter	**Summer**	**Winter**
Total COD, mg/l	297	244
Filtered COD, mg/l	121	85
*BSCOD, mg/l	80	55
Total N, mg/l	23.0	23.2
NH_4–N, mg/l	17.0	11.2
SS, mg/l	112	83
O_2, mg/l	2.7	7.5
pH	7.3	7.2
Alkalinity, mmol/l	3.0	2.2
Temperature,°C	14.7	5.5

*BSCOD=Biodegradable Soluble COD

In the winter, only 1/3 of the primary effluent COD was soluble (as defined by a 1.2 µm glass fiber filter), and only 20–25% of the COD was both soluble and biodegradable. Less than 50% of the primary effluent nitrogen was present as NH_4–N. Large differences between week-days and weekends were also observed; with biodegradable, soluble COD (BSCOD) concentrations on weekends being only about 50% of the concentrations on week-days. Primary effluent dissolved oxygen (DO) concentrations were also very high during the winter. Note the high DO concentration in the influent, especially in winter, caused by melting snow-water intrusion in the wastewater network.

3.1.5.1 Temperature dependency on nitrification.

Multiple linear regression resulted in an apparent temperature coefficient of θ = 1.07 in the Arrhenius equation ($r_{N,T2} = r_{N,T1} \cdot \theta^{(T2--1)}$), equivalent to a *real* temperature coefficient of θ=1.09. The apparent temperature coefficient is what is observed directly. The real temperature coefficient is based on a situation with a constant DO concentration in the reactor at all temperatures, and it was calculated based on the assumption that the reactor DO concentration varied in proportion with the saturation concentration at the actual temperature.

3.1.5.2 DO concentration effects on nitrification..

Data from a six-day period with some variations in DO concentrations, while other important parameters were almost constant, were used to produce Figure 3.19. The nitrification rates were calculated based on flow proportional wastewater samples collected from 8:00 am to 8:00 am the following morning. The DO concentrations were taken from on-line measurements, stored in the control system as historic mean values from midnight to midnight. The discrepancy in timing will create some scatter in the results.

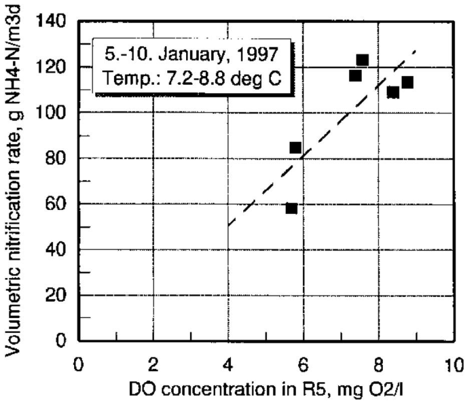


Figure 3.19. Influence of reactor DO concentration on the nitrification rate (100 g NH_4-N/m^3d equals 0.3 NH_4-N/m^2d).

However, Figure 3.19 clearly shows that increased DO concentrations resulted in significantly increased nitrification rates. The broken line indicates a 70 % increase in nitrification rate by increasing the DO concentration from 5 to 8 mg O_2/l. This confirms the possibility to control the nitrification rate in biofilm reactors by regulating the DO concentration.

3.1.5.3 Effect of primary treatment on nitrification.

For a period of 10 days the COD and SS loads on the pilot plant were doubled, by taking the primary clarifier out of service. This immediately reduced the nitrification rate in the first aerobic reactor (R3) by 50%, but had no effect on the third aerobic reactor (R5) within this 10-day period. Shortly after putting the primary treatment back on line, the first aerobic reactor (R3) nitrification rates climbed back to the original values. In the third aerobic reactor (R5), however, nitrification rates dropped by approximately 25% for about a week, in spite of an increase in reactor DO concentration. A possible explanation for this may be that excess amounts of heterotrophic biomass grown in the first and second aerobic reactors due to the previously increased organic load, were detached from these reactors and carried over to the third aerobic reactor. Some of this heterotrophic biomass may have been adsorbed to the biofilm in the third aerobic reactor and thus temporarily reduced the nitrification capacity of this reactor.

3.1.5.4 Effects of pH and alkalinity on nitrification.

Based on previous test results (Rusten *et al.*, 1995b), alkalinity and pH values down to 0.5 mmol/l and pH 6.1 in the second and third aerobic reactors were too low to achieve optimum nitrification. For the final weeks of the pilot test, however, the pH was adjusted to an average of pH 7.1 by addition of NaOH. This resulted in a 30% increase in the average nitrification rate.

3.1.5.5 Proposed design nitrification rate.

The FREVAR wastewater had a low NH_4–N/TKN ratio (TKN = total Kjeldahl nitrogen). Thus, nitrification rates measured as removed NH_4–N, were artificially low due to simultaneous hydrolysis of organic N to NH_4–N. Based on the sum of produced NO_x–N and assimilated NH_4–N, the average nitrification rate over all the aerobic reactors was 50% higher than based on influent and effluent NH_4–N concentrations. An evaluation of all the data indicated that a design nitrification rate of 190 g

TKN/m^{3}d (equal to 0.58 g TKN/m^{2}d) would be reasonable at a temperature of 10 °C, provided that the pH in the reactors is adjusted to pH=7-0. This is very close to the present value in the revised Norwegian design guideline for nitrification after pre-settling and pre-denitrification (0.6 g NH_4-N/m^{2}d at 10 °C).

3.1.5.6 Influence of oxygen on denitrification..

Figure 20 shows how the DO concentration in the reactor influenced the pre-denitrification rate.

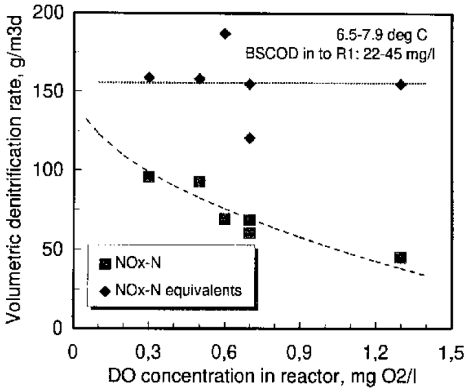


Figure 3.20. Effect of reactor DO concentrations on denitrification rates in the first anoxic reactor (150 g NH_4-N/m^{3}d equals 0.45 NO_x-N_{equiv}/m^{2}d).

Data from a period with an average wastewater temperature of 7.2 °C were used, because fairly high DO concentrations in the reactor were only observed at low temperatures. The rates given as NO_x–$N_{equivalents}$ (where removal of 1 g O_2 is equal to removal of 0.35 g NO_3–N) were almost constant, with an average value of 155 g NO_x-$N_{equivalents}$/m^{3}d (0.47 g NO_x-$N_{equiv.}$/m^{3}d) at around 7 °C . This is close to the present value in the revised Norwegian design guideline for denitrification after pre-settling (0.5 g NH_4-N/m^{2}d) at 10 °C.

The DO effect on the denitrification rate, given as g NO_x–N/m^{3}d, could be described by following equation that provided the best fit, with a correlation coefficient of r=0.93:

$$\boldsymbol{r_{DN} = r_{DN,Max}\,(1–0.66 \cdot (DO)^{1/2})}$$

r_{DN} = denitrification rate
$r_{DN,Max}$ = maximum denitrification rate
DO = oxygen concentration in reactor in mg O_2/l.

At a reactor concentration of 0.3 mg O_2/l, the above equation shows that the pre-denitrification rate will be reduced to 65% of its maximum value.

3.1.5.7 Influence of organic matter.

The denitrification rates were strongly and linearly influenced by the BSCOD concentrations (see Figure 3.21). At 10°C the values is expected to be around 10 % lower and 20 % lower at 8 °C than measured here.

With low influent BSCOD concentrations, the C/N-consumption ratios, measured as g BSCOD/g NO_x–N, were lower than with high influent BSCOD concentrations. This indicates that hydrolysis products from particulate COD were used as carbon source when available BSCOD was limited, a finding that was confirmed in several batch denitrification tests.

The denitrification rates were higher on weekdays than on weekends, due to higher influent BSCOD concentrations on week-days. The average consumption of filtered COD is shown in Table 3.4 as averages over a period of 4 months.

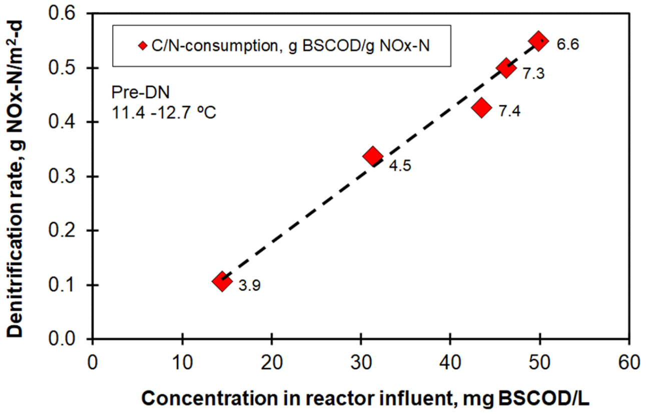


Figure 3.21. Denitrification rates and corresponding /N-consumption ratios as a function of BSCOD concentrations in the reactor influent. Data are for reactor 1 at the end of May, 1996, with 1.1 mg NO_x–N/l in the reactor.

Table 3.4 C/N-consumption ratios, with corresponding C/N ratios in primary effluent and BSCOD concentrations in to each reactor. Averages for a 4 month period with a temperature of 9.4 °C

	R1 Week-days	**R1 Weekends**	**R2**
C/N in primary effluent, g BSCOD/g TN	2.8	1.7	—
BSCOD in influent to reactor, mg/l	26	12	7
C/N-consumption, g FCOD/g NO_x–N removed	13.2	8.4	2.0
C/N-consumption, g FCOD/g NO_x–$N_{equivalents}$ removed	7.9	4.5	1.5

The large difference between ratios based on NO_x–N and ratios based on NO_x–$N_{equivalents}$, is due to the high concentrations of oxygen in the influent to reactor 1. Due to more available BSCOD, the C/N-consumption ratios were higher on week-days than on weekends. A larger fraction of hydrolysis products from particulate COD were used on weekends, resulting in a lower consumption ratio of filtered COD. Most of the COD used for denitrification in reactor 2 was taken from hydrolysis products. The reactor 2 effluent contained approximately 5 mg BSCOD/l, indicating that not all the BSCOD was easily biodegradable under anoxic conditions. This BSCOD was, however, rapidly consumed in the following aerobic reactors.

The amount of carbon needed for denitrification is temperature dependent as well, due to increased sludge yield at lower temperatures. Previous tests (Rusten et al., 1996) showed a 40–50% increase in carbon demand when the temperature was reduced from 15 to 5 °C.

3.1.5.8 Temperature dependency on denitrification..

Based on data in the temperature range from 5 to 17 °C, a true temperature coefficient of $\theta = 1.05$ was found. In reality, removal of NO_x–N will be further reduced at low temperatures due to an increased supply of oxygen. Taking into account the increased supply of oxygen in the raw water at low temperatures at FREVAR, the apparent temperature coefficient was as high as $\theta = 1.10$.

3.1.5.9 Proposed design denitrification rates.

The design denitrification rates proposed for FREVAR (at 10 °C) varied from 135 g NO_x-$N_{removed}$/$m^{3}\cdot$d (0.4 g NO_x-$N_{removed}$/$m^{2}\cdot$d) at 50 mg BSCOD/l in influent to 80 g NO_x-$N_{removed}$/$m^{3}\cdot$d (0.24 g NO_x-$N_{removed}$/$m^{2}\cdot$d) at 30 mg BSCOD/l and 20 g NO_x-$N_{removed}$/$m^{3}\cdot$d (0.06 g NO_x-$N_{removed}$/$m^{2}\cdot$d) at 10 mg BSCOD/l in the influent. The corresponding proposed values at 5 °C were 85 (0.26), 50 (0.15) and (0.03) g NO_x-$N_{removed}$/$m^{3}\cdot$d (g NO_x-$N_{removed}$/$m^{2}\cdot$d) at 50, 30 and 10 mg BSCOD/l in the influent respectively.

The rates at low temperatures are very low, due to the very high influent oxygen concentrations at FREVAR at these temperatures. If influent oxygen concentrations had been lower, higher rates would be expected. The proposed pre-denitrification rates indicate, however, that only marginal pre-denitrification will be achieved for a condition with both low temperature, low concentration of BSCOD and aerobic inlet water. Hence, for plants that may encounter such situations, it is better to safeguard design by adding a post-denitrification step.

3.2 Experiences from USA

3.2.1 Upgrade of Palmer WWTP in Alaska

Figdore et al (2019) reported on an upgrade of the wastewater treatment plant for the Alaska City of Palmer that serves a population of approximately 7,200. The effluent must meet conventional secondary treatment limits as well as seasonal monthly average and maximum daily ammonium limits of 1.7 and 3.6 mg/l in summer and 8.7 and 18.5 mg/l in winter. Prior to 2018, secondary treatment was based solely on covered aerated lagoons that were not able to reliably meet effluent ammonia limits. Nitrification was particularly challenging in Palmer's cold winter climate with influent temperatures near 8°C and further temperature decreases to near 1°C across the lagoons.

A new MBBR-plant was designed and constructed for a minimum temperature of 5°C, the average recorded 60-day wastewater temperature during performance testing was, however, 8.8°C, with minimum and maximum average daily recorded temperatures of 8.3°C and 10.5°C. Dissolved oxygen concentrations were above 5.0 mg/l.

The MBBR influent flow and TKN concentration and effluent NH_3-N concentration during the performance period of the winter of 2019 are shown in Figure 3.22. Melting of ice and snow during the month of March led to average daily flows peaks that caused a few spikes in effluent NH_3-N up to 4.0 mg/l. As soon as the flows dropped back to normal, the system recovered to very low MBBR effluent NH_3-N concentrations. MBBR effluent NH_3-N concentration averaged 0.4 and 0.9 mg/l in the first and second 30-day intervals of the performance-testing period, which met the MBBR specification requirement of less than 1.0 mg/l for a 30-day average. These values are well below the winter season permit monthly average and maximum day limits of 8.7 and 18.5 mg/l.

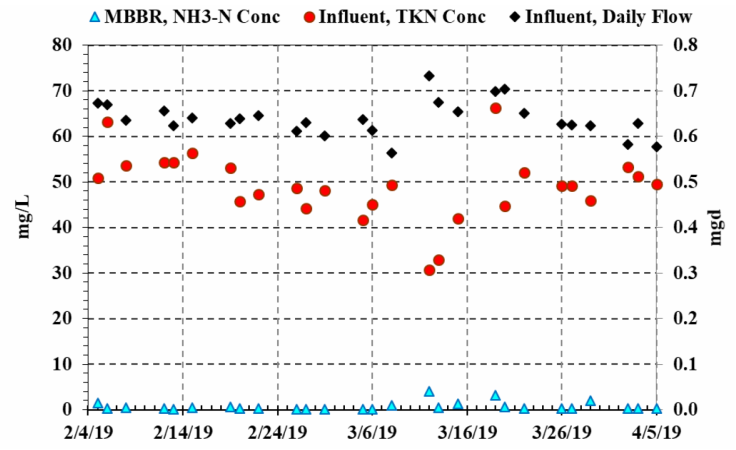


Figure 3.22. MBBR influent flow and TKN concentration and effluent NH_3-N concentration during performance verification period in the winter of 2019 at Palmer WWTP in Alaska.

3.2.2 Johnstown WWTP - Nitrification of lagoon effluent.

Wessman and Johnson (2006) reported from a full-scale plant (Johnstown WWTP) were MBBR in two parallel lines were placed between lagoon 2 and 3 in order to enhance nitrification (see Figure 3.23).

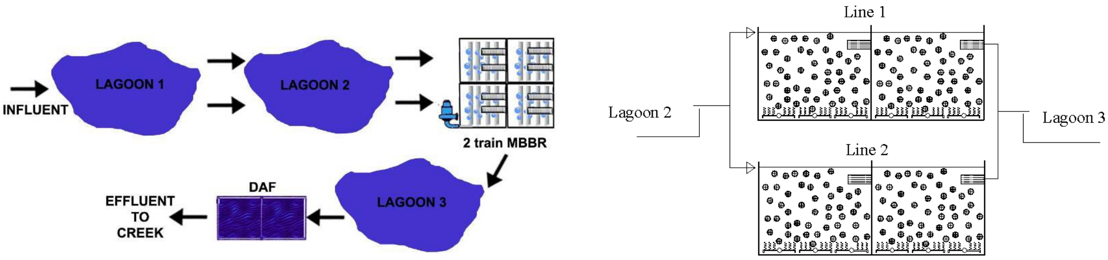


Figure 3.23. Flow diagram of Johnstown WWTP (Wessman and Johnson, 2006).

From December 2004 to mid-January 2005, the average influent water temperature was 4.2°C, the MBBR treatment system removed an average of over 40% of influent NH_3-N. In fact, the effluent NH_3-N concentration decreased by 35%, 42%, and 58% in December, January, and February, when compared with the same months in the winter prior to the installation of the MBBR treatment system. Throughout the warmer months, the MBBR treatment removed an average of 85% of influent NH_3-N while the effluent BOD has been, on average, less than 1 mg/l, making it very easy for the plant to meet its permit requirements month after month.

During the first winter there was a pH excursion to 9, while the wastewater temperature dropped to less than 3 °C. The paper shows how nitrification recovered in spite of the low temperature when pH was brought back to normal levels (around 7).

It was concluded that the new MBBR at Johnstown WWTP operated successfully through the first year with wastewater temperatures below 2 °C. Effluent concentrations were well below the discharge limits. Throughout the warmer months, the MBBR treatment removed an average of 85% of influent NH_3-N while the effluent BOD has been, on average, less than 1 mg/l.

3.3 Experiences from Canada (on nitrification of lagoon effluent)

In Canada, passive treatment systems, such as multi-pond lagoons, are common in northern climates where land is readily available. These systems are able to nitrify in the summer months due to their long retention time (HRT). In the winter months, wastewater temperatures can reach as low as 1°C for extended periods of operation. At this temperature, passive treatment systems are incapable of nitrification, and as such, there has been an increased interest in add-on nitrifying biofilm technologies. The moving bed biofilm reactor (MBBR) has demonstrated the potential for cold temperature nitrification. The combined efficiency and simple operation of the MBBR make it ideal as a potential upgrade technology for passive treatment systems.

MBBR-systems for this purpose has been studied extensively at Department of Civil Engineering, University of Ottawa in a research group under leadership of prof. Robert Delatolla. Most of the investigations have been performed as laboratory studies. The first studies investigated upgrade MBBR nitrification systems installed after the first or middle pond of a multi-lagoon treatment system to minimize the temperature decrease of the wastewater through the treatment process prior to entering the nitrifying MBBR system.

Delatolla et al (2009, 2010) reported a high dependence of attached growth nitrification kinetics on temperature, even when the oxygen concentration in the wastewater was high and showed that MBBR ammonia removal kinetics was significantly decreased at 4 ◦C. Delatolla et al. (2009) proposed a temperature and time dependent equation to model the attached growth nitrification rates as a function of elapsed time at 4 °C; where θ represents the Arrhenius temperature correction coefficient and t the elapsed time in days:

$$\theta = 3.81 \times 10^{-2} \ln(t) + 9.83 \times 10^{-1}$$

In Hoang et al (2014), two laboratory MBBRs were used to investigate MBBR nitrification rates at 20, 5, and 1 °C. Furthermore, the solids detached by the MBBR reactors were investigated and Arrhenius temperature correction models used to predict nitrification rates after long-term low-temperature exposure was evaluated. The nitrification rate at 5 °C was found to be 66 $\pm$ 3.9 % and 64 $\pm$ 3.7 % compared to the rate measured at 20 °C reactors 1 and 2, respectively. The nitrification rates at 1 °C, that covered a 4-month exposure period, were 0.22 and 0.21 gN/m^{2}d for the two reactors that was 18.7 $\pm$ 5.5 % and 15.7 $\pm$ 4.7 % compared to the rate at 20 °C. The quantity of solids detached from the MBBR bio-carriers was low and the mass of biofilm per carrier did not vary significantly at 20 °C compared to that after long-term exposure at 1 °C. Lastly, the temperature correction model proposed by Delatolla (2009) (see above) showed a strong correlation to the calculated ammonia removal rates relative to 20 °C following a gradual acclimatization period to cold temperatures.

The Hoang et al (2014) study was performed under laboratory conditions with reactors fed with a controlled synthetic wastewater and with a zero carbon/nitrogen (C/N) ratio. The zero carbon feed conditions minimized the potential overgrowth of the nitrifiers by heterotrophic bacteria. Hence, there was a need for research on the performance of attached growth nitrifying MBBR systems fed with real wastewater from a lagoon and conventional organic concentrations from conventional lagoon treatment systems at 1 ◦C.

Almomani et al (2014) carried out, therefore, a study using lab-scale MBBR reactors fed with incoming effluent from a full-scale lagoon. The study showed that significant average ammonia removal rates of 0.26 and 0.11 kg N/m^{3}d (corresponding to 1.04 g N/m^{2}d and 0.44 g N/m^{2}d at the filling fraction used – 50%) were achieved at 20 °C and 1 °C respectively. The increase in the ammonia removal rates with increasing temperature from 1 °C to 20 °C showed a strong correlation to an applied temperature correction coefficient model. No significant accumulation of effluent nitrite was observed at 1 °C or after being fed with synthetic wastewater; indicating that cold temperatures and transitions from real wastewater to synthetic wastewater did not stress the nitrifiers. Furthermore, the study demonstrated that changes in temperature or changes from real to synthetic wastewater did not affect the mass of biofilm attached per MBBR carrier.

In Young et al (2016) pilot-scale moving bed biofilm reactor (MBBR) was used to investigate the kinetics and biofilm response of municipal, tertiary nitrification at 1°C. The research demonstrates that significant rates of tertiary MBBR nitrification were attainable and stable for extended periods of operation at 1°C, with a maximum removal rate of 230 g N/m$^3 \cdot$ d (0.52 g N/m$^{2} \cdot$d at 55 % filling fraction of K5) at 1°C. At conventional nitrogen loading rates, low ammonia effluent concentrations below 5 mg N/L were achieved at 1°C. The biofilm thickness and dry weight biofilm mass were shown to be stable, with thickness values showing a correlation to the protein/polysaccharide ratio of the biofilm extracellular polymeric substances. Tertiary MBBR nitrification was shown to increase the effluent suspended solids concentrations by approximately 3 mg total SS/L, with 19–60% of effluent solids being removed after 30 min of settling. The settleability of the effluent solids was shown to be correlated to the nitrogen loading of the MBBR system.

In his PhD-thesis Bradley Young (Young, 2017) studied the interplay between nitrifying performance optimization and biofilm characteristics in lab- as well as in pilot scale at low temperatures. This research concluded the most important factor determining MBBR carrier selection is a combination of surface area and pore space size. Although high surface area to volume carriers are attractive, the propensity to clog at high loading rates significantly decreases the removal rates. The viability of the biomass and ammonia oxidizing bacterial communities were not significantly changed, indicating the ammonia removal rates were reduced due to loss of surface area in the clogged carriers. Operation at 1°C demonstrated significant rates of nitrification could be attained and stable for extended periods of operation. This study developed the first kinetic curve at 1°C with a maximum removal rate of 0.35 g N/m^2·d. The performance of the post carbon removal nitrifying MBBR systems were shown to be enhanced at 1°C by an increase in the viable embedded biomass as well as thicker biofilm. This effectively increased the number of viable cells present during low temperature operation, which partially compensated for the significant decrease in rate of ammonia removal per nitrifying cell.

At all studied loading rates at 1°C, the ammonia oxidizing bacteria were primarily in the family *Nitrosomonadaceae* (greater than 95 percent abundance of AOB population) and the nitrite oxidizing bacteria were primarily the genus *Nitrospira* (greater than 99 percent abundance of NOB population). In extreme low loaded conditions and as well during start-up phases Young (2017) found a high prevalence of bacteria not directly related to the nitrification process. Their presence however indicates a dynamic process with changes in microbial composition within the biofilm matrix in response to varying conditions. Change in microbial composition likely helps stabilize and maintain the biofilm matrix enhancing process robustness in the temperate climates.

Ahmed et al (2019) studied the performance of the nitrifying MBBR system at elevated municipal concentrations with exposures to low temperatures and cold-shock conditions down to 1 °C using an enhanced temperature-controlled room. A removal rate of 98.44 ± 4.69 gN·m^{-3}·d^{-1} (0.22 g N·m^{-2}·d^{-1} at 55 % fill fraction of K5) was identified as the intrinsic rate of nitrifying MBBR systems at 1 °C and was proposed as the conservative rate for low temperature design. It was found that two-MBBR in-series was a beneficial configuration.

A temperature threshold at which attached growth nitrification displayed a significant decrease in kinetics was identified between 2 °C and 4 °C. Arrhenius correction coefficients of 1.09 normally applied for low temperature nitrifying MBBR systems resulted in conservative modeled removal rates on average 21% lower than the measured rates. Thus, an Arrhenius correction coefficient of 1.049 was proposed between the temperatures of 10 °C and 4 °C and another correction coefficient of 1.149 to model rates at 1 °C. For the transition from 4 °C to 1 °C, the adjustment of a previously reported θ model (Delatolla, 2009 - see above) was proposed to account for exposure time at low temperatures. Finally, a comparison of nitrification kinetics between MBBR systems acclimatized to 1 °C and systems that are cold-shocked to 1 °C, demonstrated that shocked removal rates were 21% lower.

3.4 Experiences from Italy

Andreottola et al (2000) studied a full-scale upgrade of a RBC plant to MBBR for a touristic village in Northern Italy. The organic and ammonium loads were in the average 7.9 g COD m^{-2} d^{-1} and 0.9 g NH_4–N m^{-2} d^{-1}, and typical carbon and nitrogen removals in the MBBR at temperatures lower than 8°C were 73% and 72% respectively.

Salvetti et al (2006) studied the effect of wastewater temperature on the rate of nitrification in two pure oxygen moving-bed biofilm reactors, fed on secondary effluent from a municipal wastewater treatment plant. The first reactor (R1) was operated under ammonia-limiting conditions, while the second reactor (R2) was operated under oxygen-limiting conditions. The former showed a negligible influence of thermal changes on nitrification rates, while the latter showed a much higher dependence.

In the paper, a temperature coefficient ‘‘y’’ was defined as the actual ‘‘intrinsic’’ biological temperature coefficient, similar to the corresponding coefficient that is usually adopted for the design of activated-sludge processes. In addition, an ‘‘apparent’’ coefficient y_a was quantified independently, which was calculated according to the actual values of nitrification rates at different temperatures. The actual biological temperature coefficient ‘‘y’’, ranged between 1.086 and 1.109 (average value 1.098) under ammonia-limiting conditions, while under oxygen-limiting conditions was in the range 1.023–1.081 (average value 1.058). The apparent value y_a was near to unity (i.e. no temperature effect) under ammonia-limiting conditions, while only under oxygen-limiting conditions and at constant dissolved oxygen concentration ‘‘y_a’’ coincided with ‘‘y’’. An explanation was given that, under oxygen-limiting conditions, the specific biomass activity (i.e. the ratio of nitrification rate to biomass concentration) was strongly influenced by the combined effects of oxygen penetration through the biofilm and effluent temperature.

3.5 Experiences from Sweden

Welander and Mattiasson (2003) studied the denitrification process in a stirred lab-scale suspended carrier biofilm reactor at low temperatures (3–20 °C). The reactor was filled to 50% with Kaldnes K1 carriers. The denitrification rate showed only a rather weak dependence on the temperature, the rate at 3 °C being approximately 55% of that at 15 °C. The maximum denitrification rate obtained at 15 °C was 2.7 g NOx-N $m^{-2}_{carrier}$ d^{-1} while it was 1.5 g NOx-N $m^{-2}_{carrier}$ d^{-1} at 3 °C .

The maximum denitrification rate at 3 °C during an 8-day period was found to be constant. During the 8 days, the hydraulic retention time was approximately 1.5 h and the inlet NO_3-N concentration was 30 mg l^{-1}.

Ericsson (2011) reported a maximum full scale post-denitrification rate at Rya WWTP in Sweden of 2.2 g NO_x-N·$m^{-2}d^{-1}$ at around 10 °C, with methanol as carbon source.

4 N-removal in full-scale MBBR plants at low temperatures – Experiences from Norway

4.1 Lillehammer WWTP

The Lillehammer WWTP was upgraded to nitrogen removal in 1994. A Danish process consultant evaluated different biological nitrogen removal processes, and the MBBR process was chosen because it offered great flexibility and the lowest cost. A simplified flowsheet is shown in Figure 4.1. The MBBR has 9 reactors in series and reactors R1 through R7 can all be operated with aeration. R8 is the only reactor that will always be anoxic. The plant offers great operation flexibility and was originally operated with primary sedimentation; pre-denitrification in R1 and R2; aeration in reactors R3, R4 and R5 for removal of organic matter and nitrification; propeller mixing in R6 for oxygen depletion before returning nitrified water to R1; dosing of external carbon source and post-denitrification in R7 and R8; post-aeration in R9; and chemical precipitation of phosphorus and secondary sedimentation. However, in recent years the plant has been operated as indicated in Figure 4.1, with pre-precipitation and no pre-denitrification (aeration in reactors R1-R3). PAX-XL61NO is used for pre-precipitation, resulting in high removal of COD but limited removal of total P. Ethanol is used as external carbon source for post-denitrification and PAX-18 is used for post-precipitation.

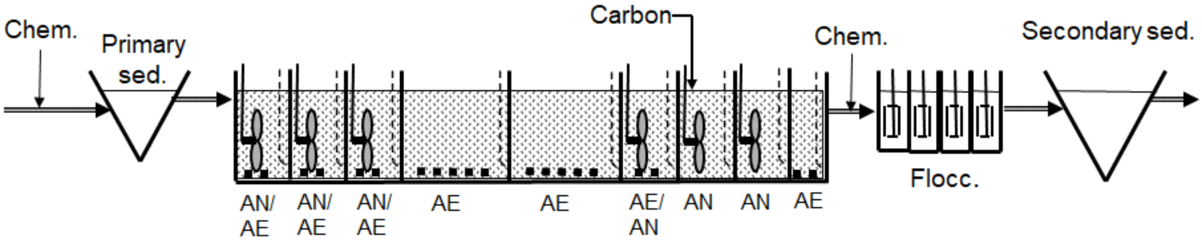


Figure 4.1. Simplified flowsheet for one train at the Lillehammer WWTP.

Table 4.1 shows design flow rates and loads for the Lillehammer WWTP.

Table 4.1. Design flow rates and loads for the Lillehammer WWTP.

		Plant influent
Design flow rates	$Q_{average}$, m³/d	26,000
	Q_{design}, m³/h	1,200
	$Q_{maxdesign}$, m³/h	1,900
Design loads	BOD_5, kg/d	2,145
	COD, kg/d	5,925
	Total N, kg/d	755
	Total P, kg/d	107

The plant has four parallel trains for enhanced primary sedimentation (pre-precipitation), with a total surface area of 600 m² and 2.5 m water depth. The MBBR has two parallel trains. Chemical precipitation of phosphorus and secondary solids removal consists of flocculation in two trains with four reactors in series and a total volume of 600 m³, and sedimentation in four trains with a total surface area of 860 m² and a water depth of 3.5 m. Table 4.2 shows reactor volumes, filling of K1 biofilm carriers and biofilm surface areas for the MBBRs. Biofilm carrier filling was measured in both MBBR trains in January 2019. The reactors were drained for water to do these measurements, so data for about 3 weeks from January 15th, 2019, will not be representative for the capacity of the treatment plant.

Table 4.2. Volumes, filling of K1 biofilm carriers and biofilm surface areas for the biological reactors at the Lillehammer WWTP in 2018 and 2019. All reactors have a water depth of 5.5 m.

Reactor	Mode	Wet volume per train, m³	K1 filling, %		Biofilm surface area* m²
			Train 1	Train 2	
R1	Aerated for BOD-removal**	180	41	24	58,091
R2	Aerated for BOD-removal**	180	39	20	53,182
R3	Aerated for BOD-removal**	180	55	9	57,273
R4	Aerated for nitrification	380	60	18	148,545
R5	Aerated for nitrification	380	45	33	148,545
R6	Aerated for nitrification, or propeller mixing for DO-reduction, or post-denitrification with external C-source	176	51	29	70,400
R7	Propeller mixing for DO-reduction, or post-denitrification with external C-source	176	45	25	62,400
R8	Post-denitrification with external C-source	176	44	13	49,600
R9	Post-aeration	92	40	31	32,618

* Total for both trains for the given reactor

** Originally designed also with propeller mixing for pre-denitrification (without aeration)

For the Lillehammer WWTP we have total COD for the influent, influent to MBBR and effluent. For total N we have only plant influent and effluent. For NH_4-N and NO_3-N we have effluent data. In order to estimate how much nitrogen that is actually nitrified and denitrified in the MBBR process, we need to estimate how much nitrogen is removed by primary sedimentation and by assimilation. We do not know how much ethanol that was added as external carbon source, but ethanol has a very low sludge yield. Using a nitrogen assimilation of 2.5 % (instead of 2.0 %) of the amount of COD entering the biological stage should more than compensate for the assimilation due to the ethanol addition.

TSS is not measured at this plant. However, the very high removal of COD over the enhanced primary sedimentation indicates removal of a lot of particulate organic material. Therefor the removal of total N over the primary sedimentation has been set at 20 % for the estimation of how much nitrogen that is removed in the MBBR process.

4.1.1 Results for Lillehammer WWTP in 2018

A summary of results for 2018 are shown in Table 4.3. Primary effluent is after pre-precipitation. The plant had an average removal of 79.7 % total N, well above the required 70 %, at an average temperature of 10.2 °C and a minimum temperature of 3.9 °C. Average removal of total P was 98.2 %. Effluent COD concentrations were always very low.

Table 4.3. Summary of results for the Lillehammer WWTP in 2018.

Parameter	Average	Median	Min.	Max.	N
Flow, m^3/d	15,212	12,399	8,972	53,928	170
Temp. in to MBBR, °C	10.2	9.8	3.9	16.2	- - -
Influent concentrations:					
COD, mg/l	702	727	166	1802	160
Total N, mg/l	54.7	58.1	21.3	88.5	153
Total P, mg/l	6.8	6.8	1.5	11.1	167
Primary effluent concentrations:					
COD, mg/l	269	248	90	657	144
Total P, mg/l	4.9	4.2	1.1	14.2	156
Effluent concentrations:					
COD, mg/l	26	25	16	47	161
Total N, mg/l	11.1	10.3	4.0	25.0	157
NH_4-N, mg/l	1.7	0.7	< 0.1	13.2	159
NO_3-N, mg/l	7.2	7.4	2.2	11.5	159
Total P, mg/l	0.12	0.11	0.02	0.46	168

Influent and effluent nitrogen concentrations, influent flow rates and water temperatures (in to the biological stage) are shown in Figure 4.2. Influent total N loads, total N discharged, influent flow rates and temperatures (in to the biological stage) are shown in Figure 4.3, together with calculated values for NH_4-N nitrified, NH_4-N discharged and NO_3-N denitrified. It was only in the winter and spring that effluent NH_4-N concentrations were high enough for nitrification to not be substrate limited. The lowest temperature dipped below 4 °C for a few days in April, when the influent flow rate increased from about 10,000 m^3/d to more than 50,000 m^3/d. Over the entire year nitrified amount was most of the time between 400 and 500 kg NH_4-N/d. Denitrified amount was typically above 300 kg NO_3-N/d at temperatures above 8 °C, but considerably lower at the lowest temperatures.

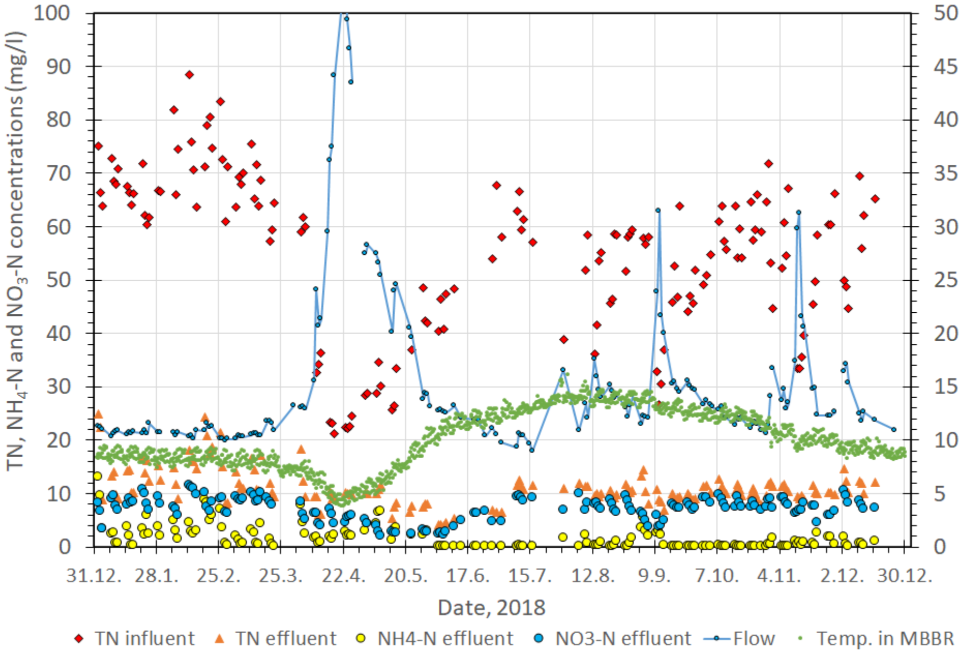


Figure 4.2. Influent and effluent nitrogen concentrations, influent flow rates and water temperatures in to the biological stage for the Lillehammer WWTP in 2018.

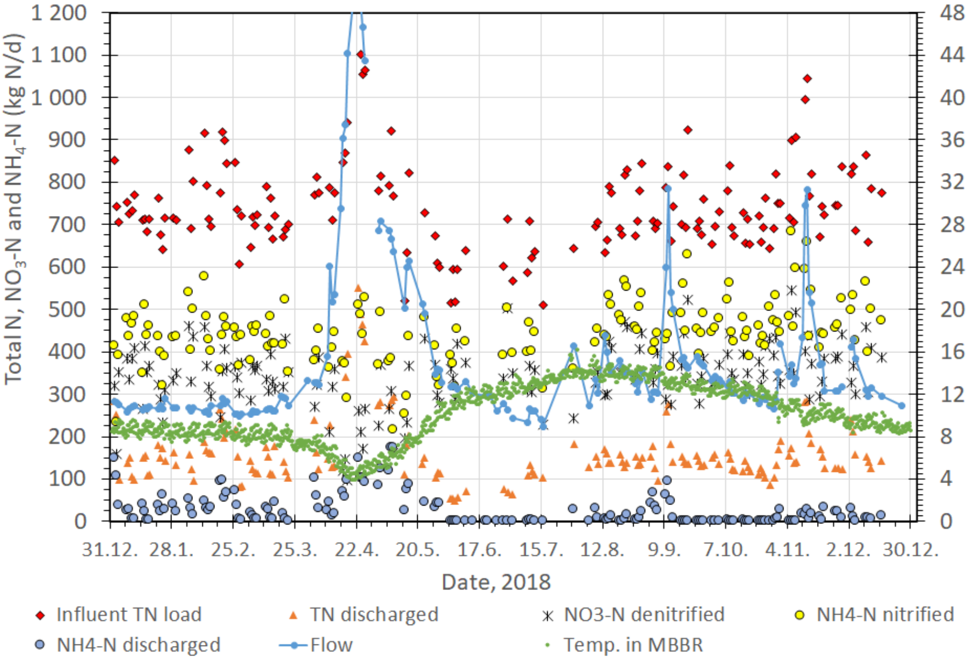


Figure 4.3. Influent total N loads, total N discharged, NH_4-N nitrified, NH_4-N discharged, NO_3-N denitrified, influent flow rates and temperatures in to the biological stage for the Lillehammer WWTP in 2018.

4.1.1.1 Specific nitrification and denitrification rates

Specific nitrification rates versus temperature are shown as weekly averages in Figure 4.4, for the period from February 5 to May 28. After this period the effluent NH_4-N concentrations were too low to show the nitrification potential. Normally a biofilm surface area equivalent to an organic load of about 5 g BOD_5/m^2-d (at 15 °C) is required for removal of organic matter, before any significant nitrification takes place (Rusten *et al.*, 1994). Figure 4.4 shows specific nitrification rates based on both the total aerobic surface area (R1 through R6) and the aerobic surface area in R4 through R6. Rates based on the total surface area will give very conservative values. Even the rates based on R4 through R6 are probably conservative, since the median organic load on R1 through R3 was as high as 25 g COD/m^2-d for the time period shown in Figure 4.4. Generally speaking, the nitrification rates were high and only marginally influenced by the temperature. We can assume that the DO in the aerobic reactors were very high in order to achieve these high nitrification rates.

Specific post-denitrification rates versus temperature are shown as weekly averages in Figure 4.5, for the period from February 5 to May 28. The graph appears to indicate a very high temperature effect. However, at temperatures below 8 °C the effluent concentrations were steady around 4-6 mg NO_3-N/l, and the significantly reduced rates at the lowest temperatures were mainly due to the large reduction in influent concentrations caused by the large increase in the influent flow rates. Since the temperature dropped due to the increased flow it is not possible to single out the effect of temperature alone. Without more information we do not know if the plant was operated to maximize the post-denitrification rates at the lowest temperatures or if the operators were happy with 4-6 mg NO_3-N/l in effluent, since the plant can easily reach the required annual average nitrogen removal.

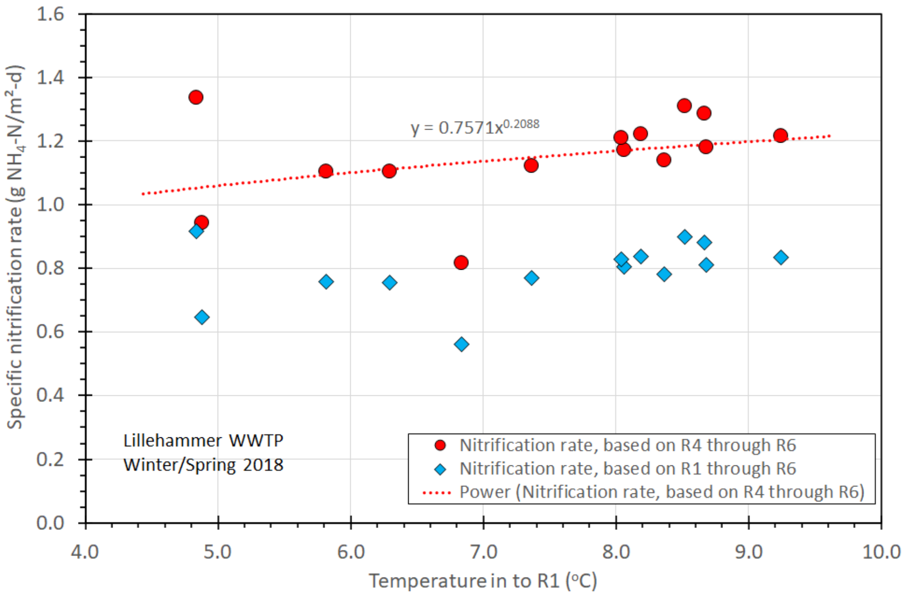


Figure 4.4. Specific nitrification rates versus temperature, based on weekly averages from February 5 to May 28, 2018.

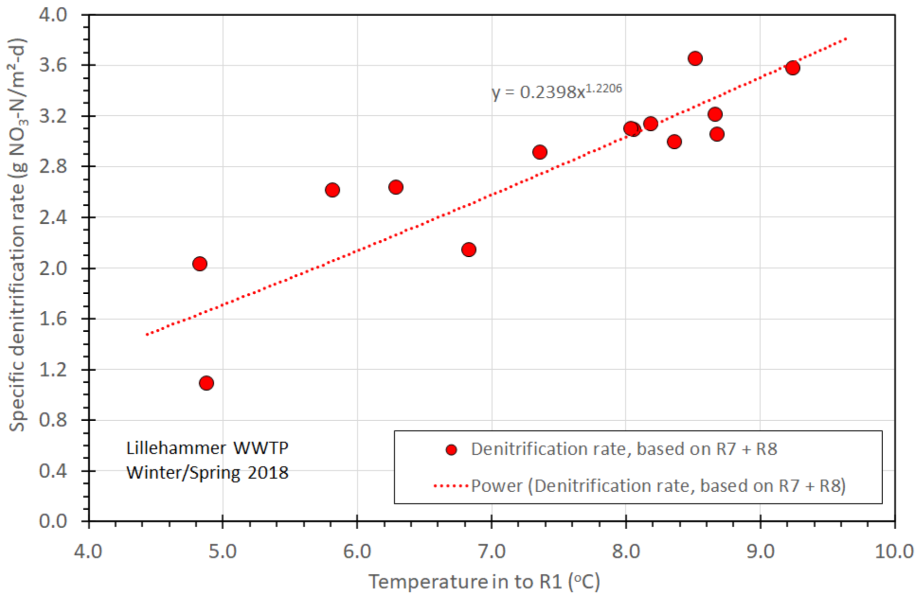


Figure 4.5. Specific post-denitrification rates versus temperature, based on weekly averages from February 5 to May 28, 2018.

4.1.2 Results for Lillehammer WWTP in 2019

A summary of results for 2019 are shown in Table 4.4. Primary effluent is after pre-precipitation. The plant had an average removal of 75.1 % total N, comfortably above the required 70 %, at an average temperature of 10.0 °C and a minimum temperature of 5.3 °C. Average removal of total P was 98.0 %. This is very good, considering the high effluent concentrations during the period when one MBBR train at a time were drained to measure the filling degree of the biofilm carriers.

Table 4.4. Summary of results for the Lillehammer WWTP in 2019.

Parameter	**Average**	**Median**	**Min.**	**Max.**	**N**
Flow, m^3/d	15,037	13,406	7,920	30,017	184
Temp. in to MBBR, °C	10.0	9.3	5.3	14.8	- - -
Influent concentrations:					
COD, mg/l	675	651	266	1,483	176
Total N, mg/l	52.6	51.1	27.4	88.5	163
Total P, mg/l	6.1	5.9	2.4	19.8	180
Primary effluent concentrations:					
COD, mg/l	624	565	238	1,297	161
Total P, mg/l	4.0	3.9	1.8	8.2	163
Effluent concentrations:					
COD, mg/l	26	25	15	70*	178
Total N, mg/l	13.1	11.6	4.7	35.2*	167
NH_4-N, mg/l	3.1	0.6	< 0.1	26.6*	169
NO_3-N, mg/l	6.9	6.6	1.3	12.2	169
Total P, mg/l	0.12	0.11	0.03	1.03*	183

* When alternate MBBR trains were drained to measure biofilm carrier filling.

Influent and effluent nitrogen concentrations, influent flow rates and water temperatures (in to the biological stage) are shown in Figure 4.6. Influent total N loads, total N discharged, influent flow rates and temperatures (in to the biological stage) are shown in Figure 4.7, together with calculated values for NH_4-N nitrified, NH_4-N discharged and NO_3-N denitrified. It was only in the winter and spring that effluent NH_4-N concentrations were high enough for nitrification to not be substrate limited. The lowest temperature dipped below 5.5 °C for a few days in April, when the influent flow rate increased from about 10,000 m³/d to about 20,000 m³/d. Over the entire year nitrified amount was most of the time between 350 and 500 kg NH_4-N/d. Denitrified amount was typically around 300 kg NO_3-N/d, with effluent concentrations of 2 -10 mg NO_3-N/l.

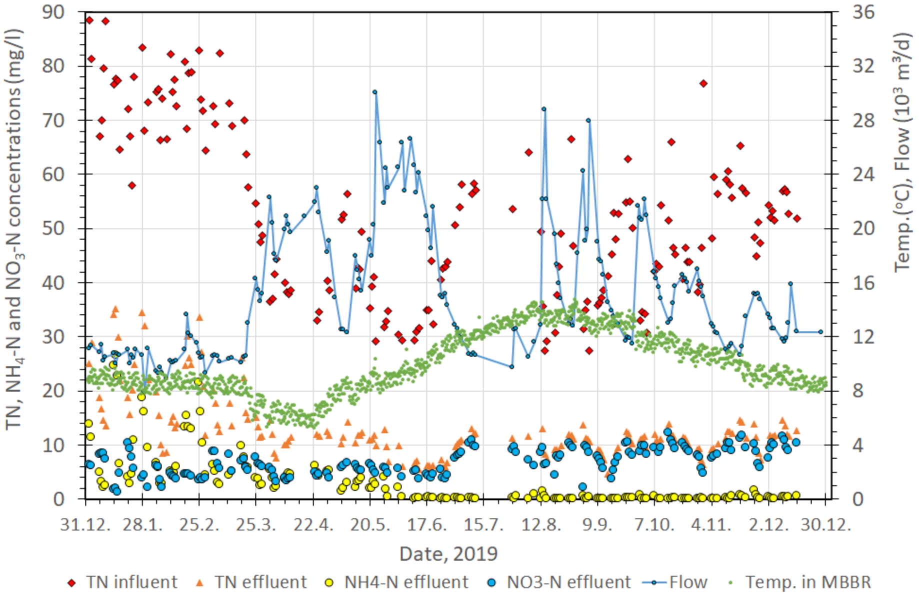


Figure 4.6. Influent and effluent nitrogen concentrations, influent flow rates and water temperatures in to the biological stage for the Lillehammer WWTP in 2019.

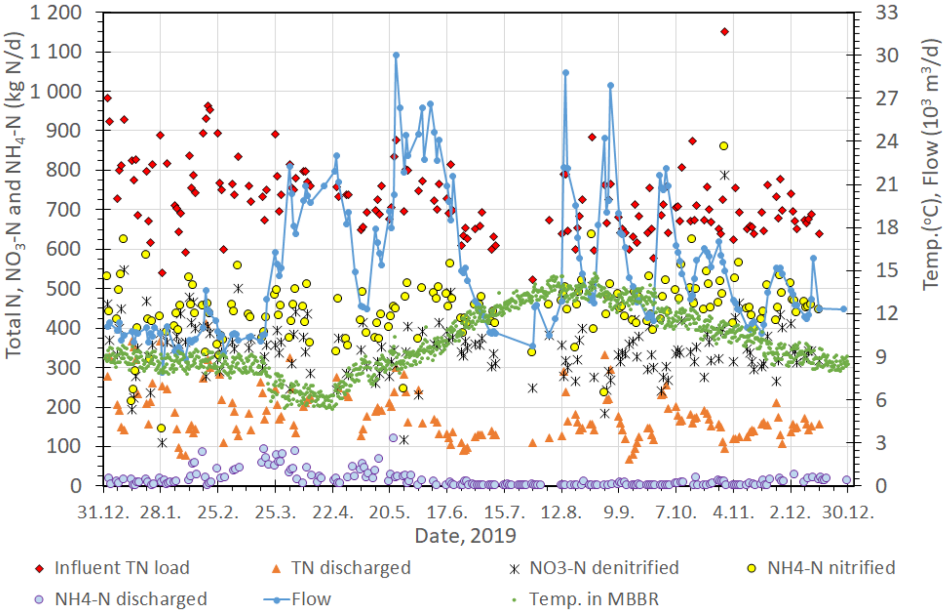


Figure 4.7. Influent total N loads, total N discharged, NH4-N nitrified, NH4-N discharged, NO3-N denitrified, influent flow rates and temperatures in to the biological stage for the Lillehammer WWTP in 2019.

4.1.2.1 Specific nitrification and denitrification rates

Specific nitrification rates versus temperature are shown as weekly averages in Figure 4.8, for the period from February 6 to June 3. After this period the effluent NH_4-N concentrations were too low to show the nitrification potential. Normally a biofilm surface area equivalent to an organic load of about 5 g BOD_5/m^2-d (at 15 °C) is required for removal of organic matter, before any significant nitrification takes place (Rusten *et al.*, 1994). Figure 4.8 shows specific nitrification rates based on both the total aerobic surface area (R1 through R6) and the aerobic surface area in R4 through R6. Rates based on the total surface area will give very conservative values. Even the rates based on R4 through R6 are probably conservative, since the median organic load on R1 through R3 was as high as 22 g COD/m^2-d for the time period shown in Figure 4.8.

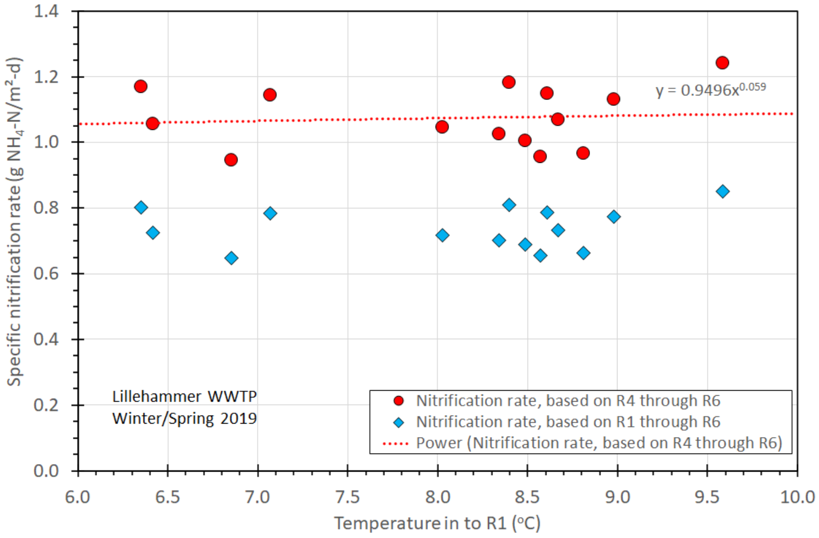


Figure 4.8. Specific nitrification rates versus temperature, based on weekly averages from February 6 to June 3, 2019.

Generally speaking, the nitrification rates were high and not visibly influenced by the temperature in the interval from 6.3 to 9.6 °C. We can assume that the DO in the aerobic reactors were very high in order to achieve these high nitrification rates.

Specific post-denitrification rates versus temperature are shown as weekly averages in Figure 4.9, for the period from February 6 to June 3. As previously shown for the nitrification rates, there was no visible temperature effect on the post-denitrification rate in the temperature interval from 6.3 to 9.6 °C. Comparing the results from 2019 with the results from 2018, it appears that the much higher flows at low temperatures in 2018 had a higher impact on the post-denitrification rates than the temperature itself.

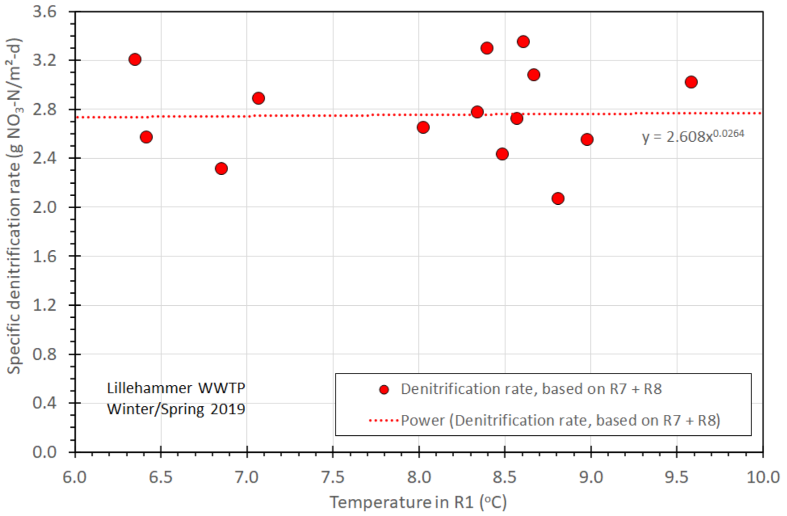


Figure 4.9. Specific post-denitrification rates versus temperature, based on weekly averages from February 6 to June 3, 2019.

4.2 Nordre Follo WWTP

The Nordre Follo WWTP is located about 20 km south of Oslo and was upgraded to nitrogen removal in 1997. Figure 4.10 shows a simplified flowsheet of the plant, while Table 4.5 shows design flow rates and loads. Pretreatment has no fat trap. Primary sedimentation has a total surface area of 696 m^2 and 2.1 m water depth. The MBBR process is followed by chemical addition for phosphorus removal, followed by flocculation (230 m^3 total volume) and dissolved air flotation (DAF) (150 m^2 total surface area). The plant receives septage and thickened sludge from small plants in the area and has anaerobic sludge digestion. This is the reason for a higher design load of total N to the biological process than the influent design load.

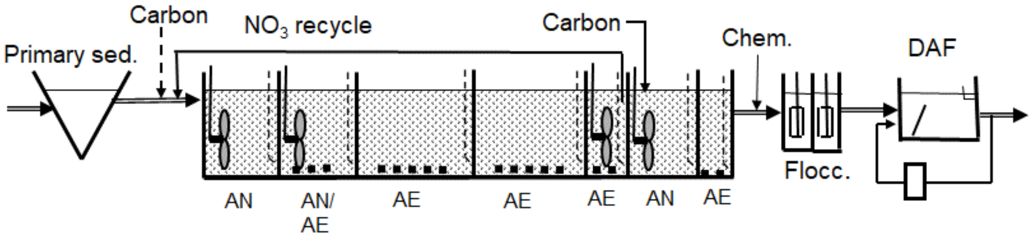


Figure 4.10. Simplified flowsheet for one of the two trains at the Nordre Follo WWTP, as operated in 2019 and 2020.

Table 4.6 shows reactor volumes, filling of K1 biofilm carriers and biofilm surface areas for each of the biological reactors in one of two identical trains. R2 can be either anoxic or aerobic, but for both 2019 and 2020 R2 was always anoxic. Historically, insufficient denitrification capacity has been the bottleneck at this plant. To improve the overall performance, operators have periodically added some methanol to the MBBR influent to boost the pre-denitrification, as indicated in Figure 4.10.

Table 4.5. Design flow rates and loads for the Nordre Follo WWTP.

		Plant influent	**Influent to biological stage**
Design flow rates	$Q_{average}$, m^3/d	14,400	- - -
	Q_{design}, m^3/h	750	- - -
	$Q_{maxdesign}$, m^3/h	1,125	- - -
Design loads (80-percentile)	BOD_5, kg/d	2,280	1,480
	COD, kg/d	5,900	3,240
	SS, kg/d	4,390	2,195
	Total N, kg/d	460	480
	Total P, kg/d	80	- - -

Table 4.6. Volumes, filling of K1 biofilm carriers and biofilm surface areas for each of the biological reactors in 2019 and 2020. Data shown for one of two identical trains at the Nordre Follo WWTP. All reactors have a water depth of 5.5 m.

Reactor	**Mode**	**Wet volume, m³**	**K1 filling, %**	**Biofilm surface area, m²**
R1	Pre-denitrification	350	52	91,000
R2	Pre-denitrification	350	49	85,750
R3	Aerated for BOD-removal and nitrification	350	58	101,500
R4	Aerated for nitrification	450	48	108,000
R5	Propeller mixing for DO-reduction	135	50	33,750
R6	Post-denitrification with external C-source	135	50	33,750
R7	Post-aeration	80	44	17,600

Due to installation of a new SCADA system at the end of 2018 temperature trends, measured in R1, are only available for 2019 and 2020.

An AnitaMox process for treatment of reject water from sludge dewatering was installed at the end of 2018 and was fully operational from April 2019. When calculating how much nitrogen that was nitrified, the amount of NO_3-N returned to the primary sedimentation tanks from the AnitaMox process has been subtracted from the total N entering the MBBR process.

We do not have total N values for the primary effluent for every day. However, for the period from August 12, 2019 to December 9, 2020, we have 72 sets of 24 h flow-proportional samples where influent total N and total N after primary treatment (influent to MBBR) were measured. These samples showed that total N concentrations in to the MBBR was on average 2.3 % higher than the total N in the plant influent, even though reject water was successfully treated (> 80 % removal of inorganic N) in the AnitaMox-process.

For calculating how much nitrogen that was nitrified and denitrified in the MBBR process, the total N in to the MBBR has conservatively been assumed to be the same as the total N in the plant influent. It has further been assumed that 20 % of the influent BOD_5 is removed by primary sedimentation and that the amount of nitrogen removed by assimilation is 4 % of the BOD_5 entering the MBBR. The assimilation of nitrogen due to the external carbon source is assumed to be 2 % of the amount of COD added.

4.2.1 Results for Nordre Follo WWTP in 2019

A summary of results for 2019 are shown in Table 4.7. The plant had an average removal of 79.8 % total N. This was well above the required 70 % total N-removal, in spite of an 80-percentile total N load that was 26 % higher than the design load. Average removal of total P was 95.7 %. Effluent BOD_5 was always ≤ 10 mg/l.

Table 4.7. Summary of results for the Nordre Follo WWTP in 2019.

Parameter	Average	Median	Min.	Max.	N
Flow, m^3/d	12,886	11,695	1,430	20,910	364
Temp. in R1, °C	11.3	11.4	3.7 *	15.6	- - -
Influent concentrations:					
BOD_5, mg/l	133	135	64	220	24
COD, mg/l	355	370	130	630	24
Total N, mg/l	43.5	42.7	16.3	86.0	361
Total P, mg/l	4.94	4.71	1.26	14.6	361
Effluent concentrations:					
BOD_5, mg/l	3.5	3.4	1.5	10	24
COD, mg/l	42	37	26	68	24
Total N, mg/l	8.8	8.3	1.6	19.9	361
NH_4-N, mg/l	1.1	0.59	< 0.1	9.3	199
NO_3-N, mg/l	6.6	6.3	3.5	13.1	210
Total P, mg/l	0.21	0.20	0.05	1.2	361
PO_4-P, mg/l	0.06	0.05	< 0.01	0.29	66
TSS, mg/l	7.1	6.8	1.2	24	204

* Most likely probe error, since this reading was in January, but at low flow. Normally low temperatures occur only in high flow situations. See Figure 4.11.

Influent and effluent nitrogen concentrations, influent flow, and water temperatures (inside R1 in the biological stage) are shown in Figure 4.11. Influent total N loads, total N discharged, influent flow rates and temperatures are shown in Figure 4.12, together with calculated values for NH_4-N nitrified and NH_4-N discharged. It was only in the winter and spring that effluent NH_4-N concentrations were high enough for nitrification to not be substrate limited. The lowest temperature measured inside reactor R1 was 6.9 °C in the second half of March. Normal winter temperatures were 9 -10 °C, but dropped to between 7 and 7.5 °C when the influent flow rate increased to the 15,000 to 20,000 m^3/d range. Over the entire year nitrified amount was most of the time between 400 and 500 kg NH_4-N/d.

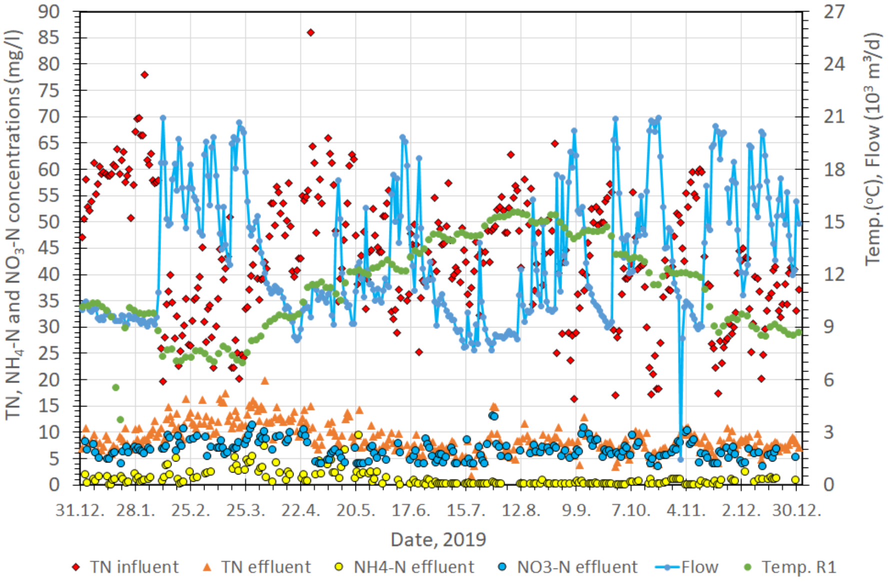


Figure 4.11. Influent and effluent nitrogen concentrations, influent flow rates and water temperatures in biological reactor R1 for the Nordre Follo WWTP in 2019.

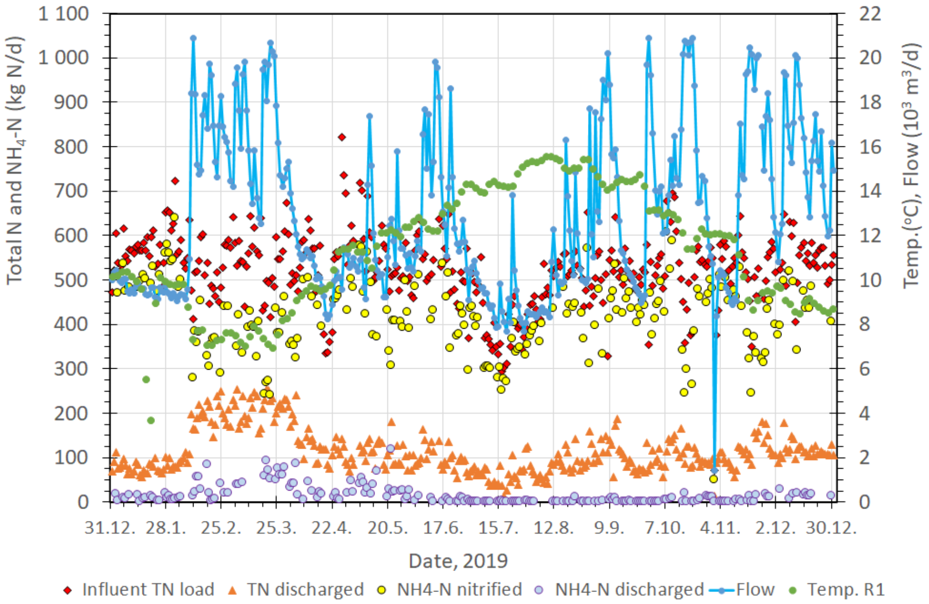


Figure 4.12. Influent total N loads, total N discharged, NH_4-N nitrified, NH_4-N discharged, influent flow rates and temperatures in biological reactor R1 for the Nordre Follo WWTP in 2019.

4.2.1.1 Specific nitrification and denitrification rates

Specific nitrification rates versus temperature are shown as weekly averages in Figure 4.13, for the period from January 28 to May 10. After this period the effluent NH_4-N concentrations were too low to show the nitrification potential. Normally a biofilm surface area equivalent to an organic load of about 5 g BOD_5/m^2-d (at 15 °C) is required for removal of organic matter, before any significant nitrification takes place (Rusten *et al.*, 1994). A significant part of the easily biodegradable organic matter may be removed over the pre-denitrification stage (R1 + R2), so it is difficult to estimate the BOD-load on R3. Figure 4.13 shows specific nitrification rates based on both the total aerobic surface area (R3 + R4 + R5) and the aerobic surface area in only R4 + R5. Rates based on the total surface area (shown with blue squares) will give conservative values, since organic matter will reduce the nitrification rate in R3. However, significant nitrification is expected to take place in R3. Thus, the nitrification rates based on the total aerobic surface area (blue squares) will be closer to the actual nitrification rate in R4 than the overly optimistic rates shown with red circles, that assume all nitrification to take place in R4 and R5. Generally, the nitrification rates (blue squares) were as expected, assuming high DO concentrations in R3 and R4. The trend line indicates a temperature coefficient (θ) for nitrification of about 1.10 in the temperature interval from 7.0 to 11.5 °C.

The overall denitrified amount from January 28 to May 10 was on average 314 kg NO_3-N/d. With the data available, it is not possible to distinguish between what happens in the pre-denitrification and in the post-denitrification. However, if we assume a reasonable post-denitrification rate of 1.5 g NO_3-N/m^2-d about 100 kg N/d is removed by post-denitrification. The mass balance will then give us an average pre-denitrification rate of 0.60 g NO_3-N/m^2-d. This fairly high pre-denitrification rate is probably boosted by methanol added to the MBBR influent, even though pre-denitrification rates in this range have been observed without external carbon addition at another MBBR plant at temperatures of 7.5 to 8.0 °C (Rusten and Ødegaard, 2007).

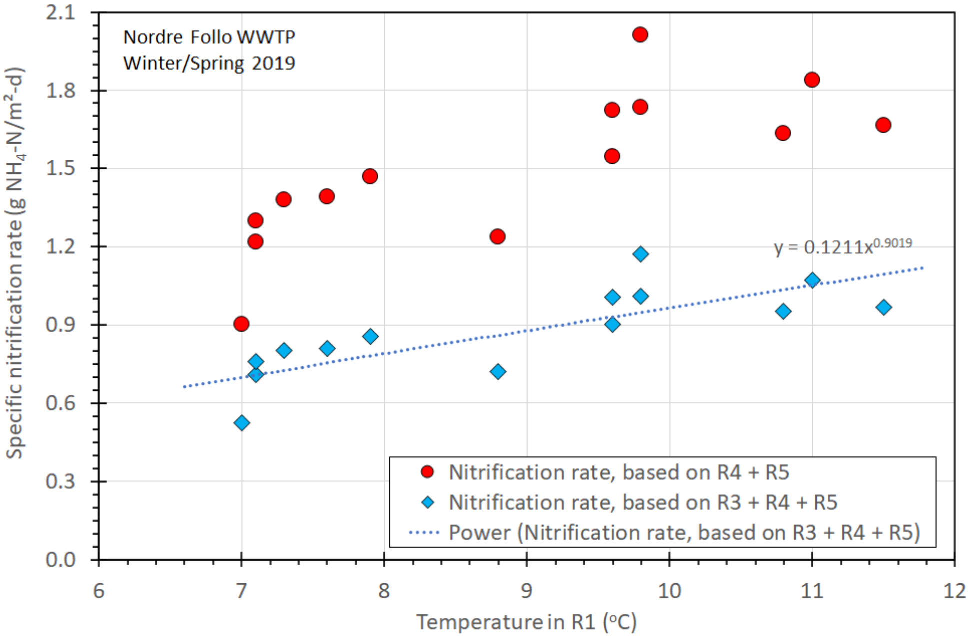


Figure 4.13. Specific nitrification rates versus temperature, based on weekly averages from January 28 to May 10, 2019.

4.2.2 Results for Nordre Follo WWTP in 2020

A summary of results for 2020 are shown in Table 4.8. The plant had an average removal of 78.5 % total N, at an average temperature of 11.8 °C and a minimum temperature of 7.5 °C. This was well above the required 70 % total N-removal, in spite of an 80-percentile total N load that was 33 % higher than the design load. Average removal of total P was 96.9 %. Effluent BOD_5 was always ≤ 4 mg/l.

Table 4.8. Summary of results for the Nordre Follo WWTP in 2020.

Parameter	Average	Median	Min.	Max.	N
Flow, m³/d	13,305	12,050	7,760	21,310	364
Temp. in R1, °C	11.8	11.7	7.5	16.0	- - -
Influent concentrations:					
BOD5, mg/l	124	110	66	200	23
COD, mg/l	459	420	230	740	23
Total N, mg/l	44.7	45.8	13.0	74.8	366
Total P, mg/l	5.16	4.98	1.23	15.0	366
Effluent concentrations:					
BOD5, mg/l	2.5	2.5	1.5	4.0	23
COD, mg/l	29	24	17	59	23
Total N, mg/l	9.6	8.9	2.1	22.0	364
NH4-N, mg/l	2.1	1.3	< 0.1	13.1	179
NO3-N, mg/l	6.5	6.3	0.26	13.2	177
Total P, mg/l	0.16	0.15	0.04	0.74	364
PO4-P, mg/l	0.06	0.05	< 0.01	0.28	36
TSS, mg/l	5.4	5.2	0.8	13	205

Influent and effluent nitrogen concentrations, influent flow rates and water temperatures (inside R1 in the biological stage) are shown in Figure 4.14. Influent total N loads, total N discharged, influent flow rates and temperatures are shown in Figure 4.15, together with calculated values for NH_4-N nitrified and NH_4-N discharged. It was only in the winter and spring that effluent NH_4-N concentration were high enough for nitrification to not be substrate limited. The lowest temperature measured inside reactor R1 was 7.5 °C in mid-February and mid-March, in connection with spikes in influent flow rates. Over the entire year nitrified amount was most of the time between 400 and 500 kg NH_4-N/d, with some reductions at the lowest temperatures and highest flow rates.

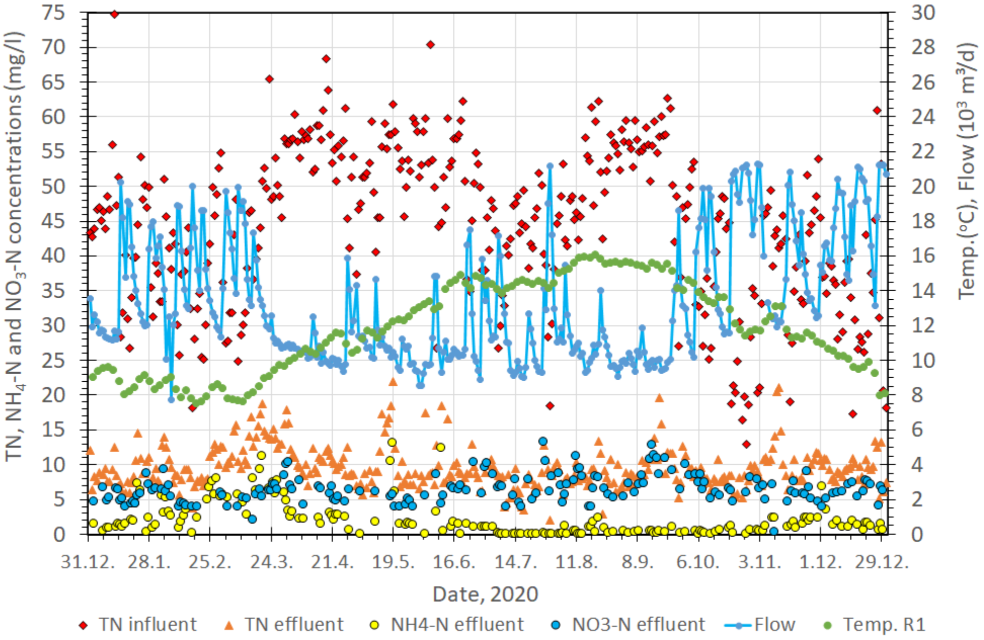


Figure 4.14. Influent and effluent nitrogen concentrations, influent flow rates and water temperatures in biological reactor R1 for the Nordre Follo WWTP in 2020.

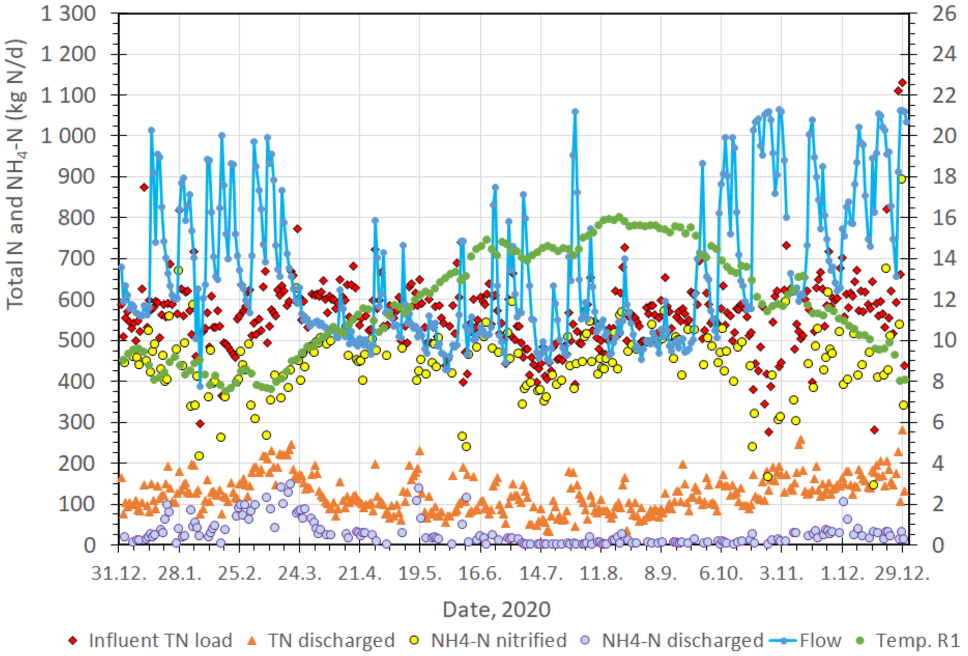


Figure 4.15. Influent total N loads, total N discharged, NH_4-N nitrified, NH_4-N discharged, influent flow rates and temperatures in biological reactor R1 for the Nordre Follo WWTP in 2020.

4.2.3.1 Specific nitrification and denitrification rates

Specific nitrification rates versus temperature are shown as weekly averages in Figure 4.16, for the period from January 13 to April 8.

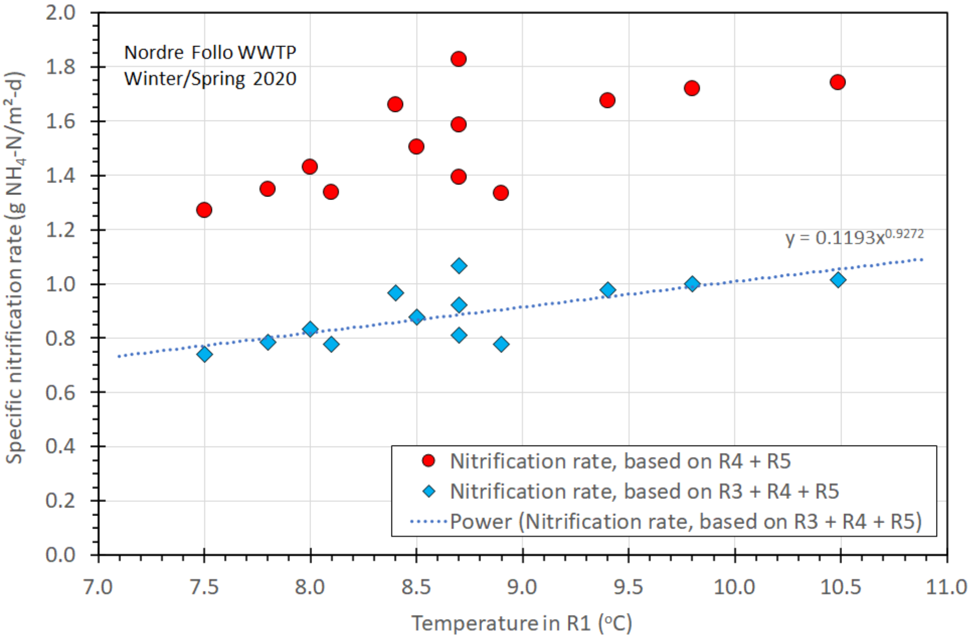


Figure 4.16. Specific nitrification rates versus temperature, based on weekly averages from January 13 to April 8, 2020. Excludes days with effluent NH_4-N below 0.2 mg/l.

After this period the effluent NH_4-N concentrations were too low to show the nitrification potential. Normally a biofilm surface area equivalent to an organic load of about 5 g BOD_5/m²-d (at 15 °C) is required for removal of organic matter, before any significant nitrification takes place (Rusten *et al.*, 1994). A significant part of the easily biodegradable organic matter may be removed over the pre-denitrification stage (R1 + R2), so it is difficult to estimate the BOD-load on R3. Figure 4.16 shows specific nitrification rates based on both the total aerobic surface area (R3 + R4 + R5) and the aerobic surface area in only R4 + R5. Rates based on the total surface area (shown with blue squares) will give conservative values, since organic matter will reduce the nitrification rate in R3. However, significant nitrification is expected to take place in R3. Thus, the nitrification rates based on the total aerobic surface area (blue squares) will be closer to the actual nitrification rate in R4 than the overly optimistic rates shown with red circles, that assume all nitrification to take place in R4 and R5. Generally, the nitrification rates (blue squares) were as expected, assuming high DO concentrations in R3 and R4. The trend line indicates a temperature coefficient (θ) for nitrification of about 1.11 from 7.5 to 10.5 °C.

The overall denitrified amount from January 13 to April 8 was on average 350 kg NO_3-N/d. With the data available, it is not possible to distinguish between what happens in the pre-denitrification and in the post-denitrification. However, about 100 kg N/d is removed by post-denitrification if we assume a reasonable post-denitrification rate of 1.5 g NO_3-N/m²-d. The mass balance will then give us an average pre-denitrification rate of 0.70 g NO_3-N/m²-d. This fairly high pre-denitrification rate is probably boosted by methanol added to the MBBR influent, even though pre-denitrification rates in this range have been observed without external carbon addition at another MBBR plant at temperatures of 7.5 to 8.0 °C (Rusten and Ødegaard, 2007).

4.3 Nedre Romerike Avløpsselskap

The Nedre Romerike Avløpsselskap IKS (NRA) WWTP is located in Strømmen, just North-East of Oslo. Figure 4.17 shows a simplified flowsheet of the plant as it was from the start-up in 2003 to 2018. During 2018 the primary sedimentation was replaced with primary filtration (Salsnes Filter channel sieves). The biological nitrogen removal process is a MBBR process with combined pre- and post-denitrification. Design flow rates and loads are shown in Table 4.9.

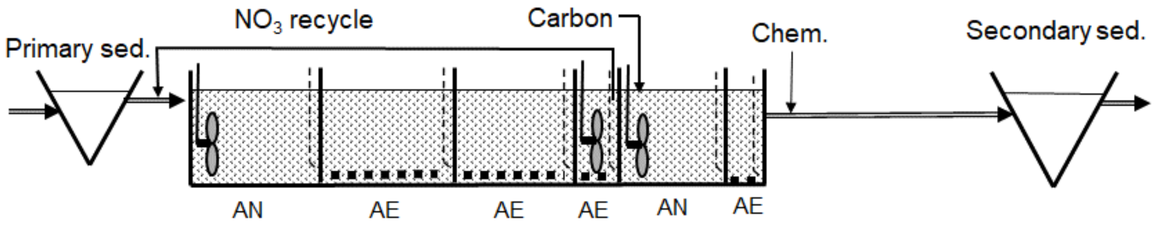


Figure 4.17. Simplified flowsheet for one of the four trains at NRA.

Table 4.9. Design flow rates and loads for NRA.

Design		**Influent**	**After primary sed.**
Design flow rates	Q_{design}, m³/h	2,300	2,300
	$Q_{maxdesign}$, m³/h	7,200	7,200
Design loads (80-percentile)	BOD_5, kg/d	8,050	6,037
	COD, kg/d	20,980	13,634
	SS, kg/d	12,990	6,490
	Total N, kg/d	1,620	1,458
	Total P, kg/d	209	188

Screens and grit chambers are used for pre-treatment. The primary sedimentation had a total surface area of 1,224 m² and 2.6 m water depth but has been replaced by 10 rotating belt filters (Salsnes SFK 600) with a total submerged filter cloth area of 22 m². Chemicals for precipitation of phosphorus are added downstream of the MBBR. There are no dedicated flocculation reactors. Flocculation takes place in the channels between the MBBRs and the secondary sedimentation tanks, which have a total surface area of 2,352 m² and a 2.6 m water depth.

The MBBRs are arranged in 4 parallel trains, each with 6 reactors in series. Kaldnes K1 biofilm carriers are used, with a protected surface area of 500 m²/m³ bulk volume of carriers. Nitrified water is returned from R4 to R1, where the organic matter in the primary treated water is used as carbon source for pre-denitrification. R4 is normally operated with only mixing, to reduce the DO concentration in the nitrified water returned back to R1 or passed on to post-denitrification in R5. If poor nitrification is observed, R4 may be aerated to boost nitrification. Methanol is normally used as external carbon source, but ethanol has been used in some trains when available in the form of illegal alcohol seized by customs. Dimensions and biofilm carrier filling for the MBBRs are shown in Table 4.10. Aerobic reactors are served by 3 HV-Turbo blowers, with variable capacity from 4,300 to 8,750 Nm³/h per unit. The biological reactors are made of glass fiber reinforced plastic, as shown in Figure 4.18.

Table 4.10. Reactor dimensions, filling of K1 biofilm carriers and biofilm surface areas for each of the biological reactors. Data shown for one of four identical trains at NRA.

Reactor	**Mode**	**Water depth, m**	**Wet volume, m³**	**K1 filling, %**	**Biofilm surface area, m²**
R1	Pre-denitrification	8.2	1,164	54	314,280
R2	Aerated for BOD-removal and nitrification	8.0	1,135	49	278,075
R3	Aerated for nitrification	7.9	1,121	49	274,645
R4	Propeller mixing for DO-reduction	7.8	211	52	54,860
R5	Post-denitrification with external C-source	7.2	1,021	14	71,470
R6	Post-aeration	7.0	190	47	44,650

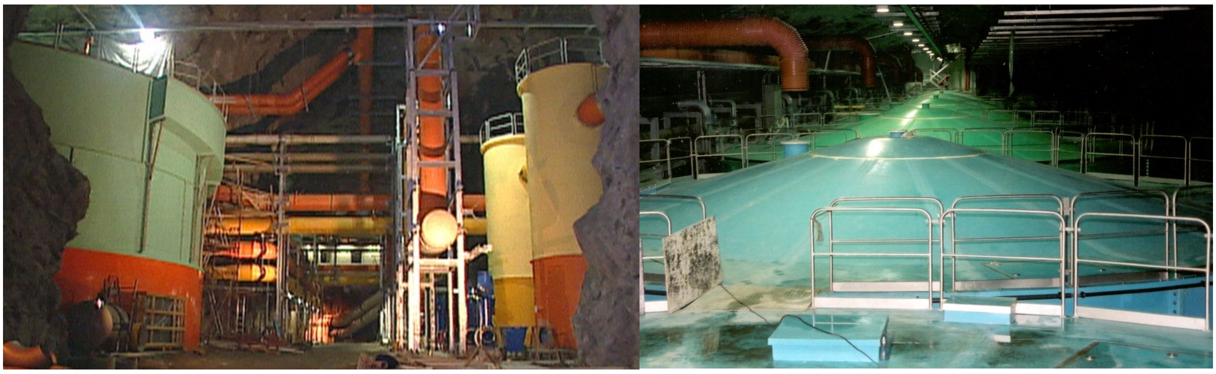


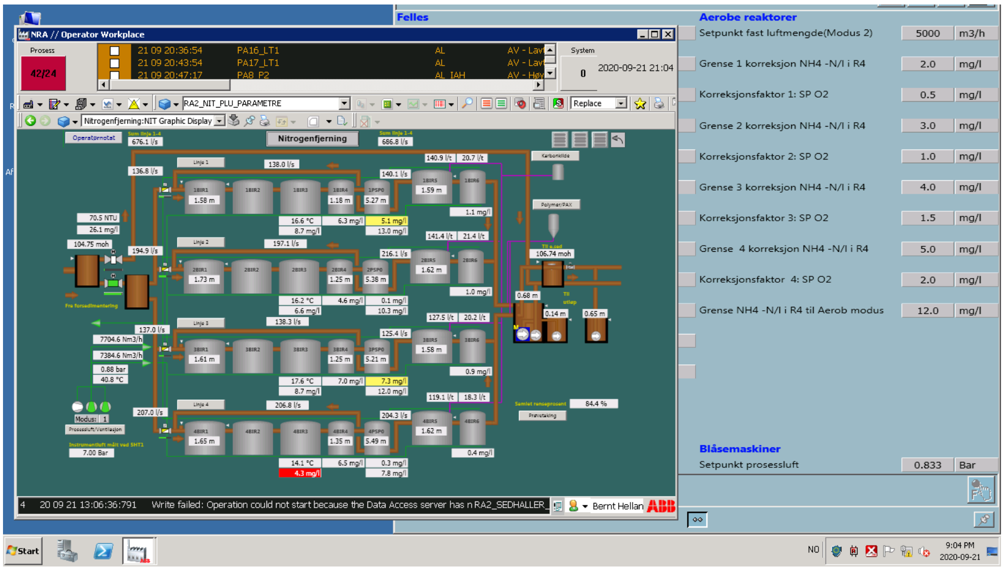
Figure 4.18. Biological reactors in glass fiber reinforced plastic (GRP) at NRA.

The biological stage is designed for a hydraulic capacity of 500 l/s per train (2,000 l/s total). The plant has been tested at this capacity, but to be conservative NRA have listed Q_{max} = 450 l/s per train in the plant specifications. This includes the flow recirculated from R4 back to R1, so when the plant is operated with pre-denitrification the capacity for influent flow to each train will be much lower.

Annually NRA take 26 flow proportional weekly (7-day) samples of plant influent and effluent, as required by the authorities (Statsforvalteren). These samples are analyzed for total BOD_5, total COD, total N and total P. TSS is not measured because the authorities assume that a spike in effluent TSS will show up in the measurement of total P. For daily operation and control of the biological process on-line measurements of DO, temperature, NH_4-N and NOx-N are used. The screen-shot in Figure 4.19 shows an example of these on-line measurements. On-line data have revealed periods with large differences in performance between the different trains.

From NRA we have received all temperature data, flow rates and analytical results of the weekly flow proportional samples for the years 2013 through 2020. We have chosen to look closer at the 4 years with the lowest temperatures over a period of 2-4 weeks, namely the years 2013, 2014, 2016 and 2018. We also have access to all the logged NH_4-N and NOx-N data for these years.

Please observe that all temperatures are measured in R3 in the biological stage and will be marginally higher than in the plant influent. There are no measurements of plant influent temperatures. The plant does not have anaerobic sludge digestion. Lime is used for sludge hygienization and stabilization.



```
21 09 20:36:54   PA16_LT1   AL       AV - Lav
21 09 20:43:54   PA17_LT1   AL       AV - Lav
21 09 20:47:17   PA8_P2     AL_IAH   AV - Høy
```

```
4   20 09 21 13:06:36:791   Write failed: Operation could not start because the Data Access server has n RA2_SEDHALLER_
```

Aerobe reaktorer

Setpunkt fast luftmengde(Modus 2)	5000	m3/h
Grense 1 korreksjon NH4 -N/l i R4	2.0	mg/l
Korreksjonsfaktor 1: SP O2	0.5	mg/l
Grense 2 korreksjon NH4 -N/l i R4	3.0	mg/l
Korreksjonsfaktor 2: SP O2	1.0	mg/l
Grense 3 korreksjon NH4 -N/l i R4	4.0	mg/l
Korreksjonsfaktor 3: SP O2	1.5	mg/l
Grense 4 korreksjon NH4 -N/l i R4	5.0	mg/l
Korreksjonsfaktor 4: SP O2	2.0	mg/l
Grense NH4 -N/l i R4 til Aerob modus	12.0	mg/l

Blåsemaskiner

Setpunkt prosessluft	0.833	Bar

Figure 4.19. Example of screen-shot with on-line measurements in the MBBR stage at the NRA WWTP.

4.3.1 Results for NRA in 2013

Figure 4.20 shows flow through the MBBR process and average water temperature in R3 for the 4 trains at NRA for every day of the year. Plant influent and effluent total N concentrations are also shown, based on 26 sets of weekly flow proportional samples. Flows that by-passed the MBBR stage are not shown, but at flows above approximately 80,000 m³/d through the MBBRs significant amounts of water were by-passed. The lowest water temperature was 5.2 °C on April 13.

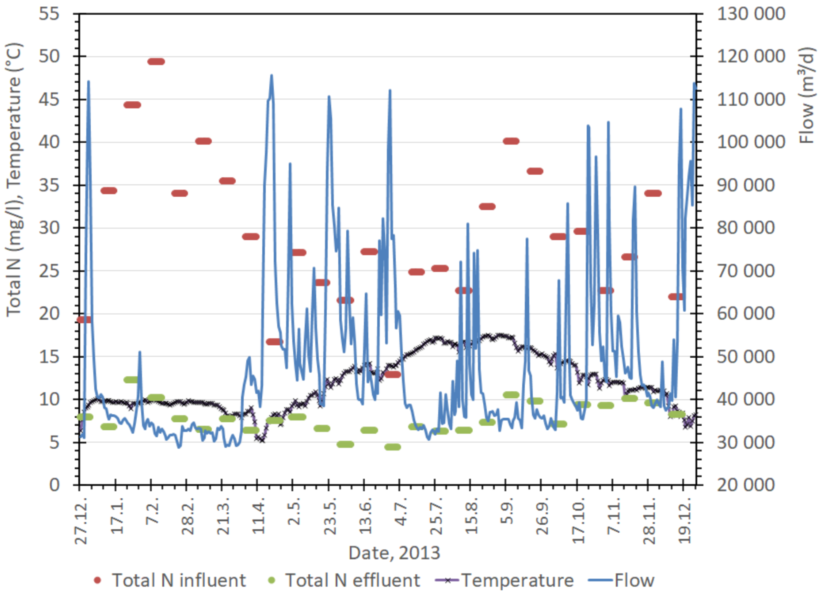


Figure 4.20. Flow through the MBBR process and average water temperature in R3 for the 4 trains at NRA for every day of the year 2013. Plant influent and effluent total N concentrations in weekly flow proportional samples are also shown.

For the part of the year with the lowest water temperatures additional results are shown in Table 4.11, as averages for the given week. This will typically be in connection with melting snow and precipitation in the spring. Removal of total N was reduced from 78 % at 8.3 °C in week 15 to 55 % at 6.6 °C in week 17. There was only a minor difference in effluent total N concentrations in these two weeks, but due to very high flows and a lot of water in by-pass the influent concentrations were low in week 17. Removed amount was 1,027 kg N/d in week 15, and dropped to 763 kg N/d in week 17.

The on-line measurements shown in Table 4.11 are strict averages over the given week and not flow proportional. However, these on-line data can be used to estimate specific nitrification rates and denitrification rates. Nitrification rates are calculated based on NH_4-N concentrations in the primary effluent and out of R4, using the total biofilm surface area in R2 + R3 + R4 and the average weekly flow rate. Pre denitrification rates are calculated based on primary effluent NH_4-N concentrations, minus NH_4-N and NOx-N concentrations in R4, in addition to the biofilm surface area in R1 and the average weekly flow rate. These specific reaction rates will be conservative because they will not include the amount of organic N hydrolyzed in the reactors and subsequently nitrified and denitrified, nor will they include any NOx-N in the influent which is typically seen during storm water events. Furthermore, nitrification rates measured over R2 + R3 + R4 will be low because nitrification in R2 will be heavily influenced by the amount of biodegradable organic matter coming from R1. Post-denitrification rates are calculated based on the measured NOx-N concentrations out of R4 and R6, plus the biofilm surface area in R5 and the average weekly flow rate. The calculated post-denitrification rates will be fairly accurate, but marginally lower than the real rates if any residual NH_4-N is nitrified during post-aeration in R6.

Calculated post-denitrification rates showed no pattern of temperature dependency. The average rate over all these weeks was 1.11 g NOx-N/m²-d, identical to the rate in week 17 at the lowest weekly temperature of 6.6 °C. However, pre-denitrification rates were low and showed very large variations. In week 15 the calculated pre-denitrification rate was 0.27 g NOx-N/m²-d. The rate dropped to only 0.025 g NOx-N/m²-d in week 17, most likely due to the very high flow and underestimation of the real rate due to influent organic N and NOx-N. In week 19, when the flow was similar to the flow in week 15, the calculated pre-denitrification rate increased to 0.32 g NOx-N/m²-d.

Calculated nitrification rates over R2 + R3 + R4 varied from 0.30 to 0.40 g NH_4-N/m²-d for the weeks shown in Table 4.11. The average rate was 0.36 g NH_4-N/m²-d and no temperature dependency was seen (plotting rates versus temperature the trend line showed a very slight increase in rate at the lower temperatures).

The results indicate that high flow and dilute wastewater were more important for the reduced removal efficiency of total N in week 17 than the reduced temperature. The role of dilute wastewater can also be seen by going back to Figure 4.20 and compare week 45 (31/10 – 6/11) and week 49 (28/11 – 4/12), even though the differences in influent concentrations were not as large as between week 15 and week 17. In week 45, with a flow of 62,470 m³/d and 12.4 °C, removal of total N was 59 %. In week 49 the flow was reduced to 39,568 m³/d, and even though the temperature dropped to 11.4 °C the removal efficiency increased to 72 % for total N.

Table 4.11. Temperatures, flow rates, concentrations and nitrogen loads for NRA in the spring of 2013. Also shown are on-line NH_4-N and NOx-N measurements (as weekly averages), plus calculated specific nitrification and denitrification rates based on these on-line measurements.

Parameter	Week and date, 2013					
	11 7/3- 13/3	**13 21/3- 27/3**	**15 4/4- 10/4**	**17* 18/4- 24/4**	**19 2/5- 8/5**	**21* 16/5- 22/5**
Temperature, °C	9.7	9.1	8.3	6.6	9.0	10.4
Flow rate, m^3/d	32166	30602	45605	83160	48894	52237
Load:						
Total N, kg/d	1290	1083	1318	1389	1325	1233
Influent concentrations:						
BOD_5, mg/l	220	230	230	100	170	140
COD, mg/l	640	600	490	260	490	320
Total N, mg/l	40.1	35.4	28.9	16.7	27.1	23.6
Total P, mg/l	4.84	4.86	3.56	1.74	3.36	3.22
Effluent concentrations, mg/l						
BOD_5, mg/l	1.5	3	1.5	19	4	1.5
COD, mg/l	40	15	40	50	15	41
Total N, mg/l	6.49	7.66	6.40	7.53	7.87	6.61
Total P, mg/l	0.20	0.19	0.12	0.14	0.17	0.14
Removal efficiencies:						
BOD_5, %	99.3	98.7	99.3	81.0	97.6	98.9
COD, %	93.8	97.5	91.8	80.8	96.9	87.2
Total N, %	83.8	78.4	77.9	54.9	71.0	72.0
Total P, %	95.9	96.1	96.6	92.0	94.9	95.7
On-line measurements:						
Primary effluent NH_4-N, mg/l	25.8	26.4	23.3	13.2	23.22	19.6
R4 effluent NH_4-N, mg/l	2.95	1.93	2.44	2.79	3.14	1.47
R4 effluent NOx-N, mg/l	20.0	18.1	13.4	9.99	11.8	12.2
R6 effluent NOx-N, mg/l	8.87	9.05	5.90	6.17	5.20	6.62
Rates calculated from on-line data:						
Nitrification rate, g NH_4-N/m-d **	0.303	0.308	0.392	0.354	0.404	0.389
Pre-DN rate, g NOx-N/m^2-d §	0.074	0.156	0.272	0.025	0.320	0.247
Post-DN rate, g NOx-N/m^2-d §§	1.25	0.968	1.20	1.11	1.13	1.01

* Water in by-pass. Given flows, loads and effluent concentrations are for water that has been through the entire treatment plant.

** Based on total biofilm surface area in R2 + R3 + R4.

§ Based on biofilm surface area in R1.

§§ Based on biofilm surface area in R5.

4.3.2 Results for NRA in 2014

Figure 4.21 shows flow through the MBBR process and average water temperature in R3 for the 4 trains at NRA for every day of the year. Plant influent and effluent total N concentrations are also shown, based on 26 sets of weekly flow proportional samples. Flows that by-passed the MBBR stage are not shown, but for some flows above approximately 80,000 m^3/d through the MBBRs, and for all flows above approximately 110,000 to 115,000 m^3/d, significant amounts of water were by-passed. The lowest water temperature was 5.0 °C on February 16.

For the part of the year with the lowest water temperatures additional results are shown in Table 4.12, as averages for the given week. This year (2014) had very high flow rates and low temperatures early in the year, so Table 4.12 shows data from week 4 through week 18. Removal of total N was 73 % in week 4, with the very high load of 1,741 kg TN/d, a water temperature of 9.6 °C and a flow of 41,352 m^3/d. However, weeks 8, 10 and 11 had extremely high flows, weekly average temperatures from 6.4 to 7.2 °C, and also extremely low influent concentrations of BOD_5 and total N. Removal efficiencies with respect to total N were only from 35 to 46 % during these weeks. In week 14, with significantly reduced flow, the total N removal efficiency increased to 75 %. However, the effluent total N concentrations were very stable and showed only minor variations from week 8 through week 18. The amount of total N removed in the treatment plant was as high as 1269 kg N/d in week 4, while only 455 kg N/d was removed in week 8.

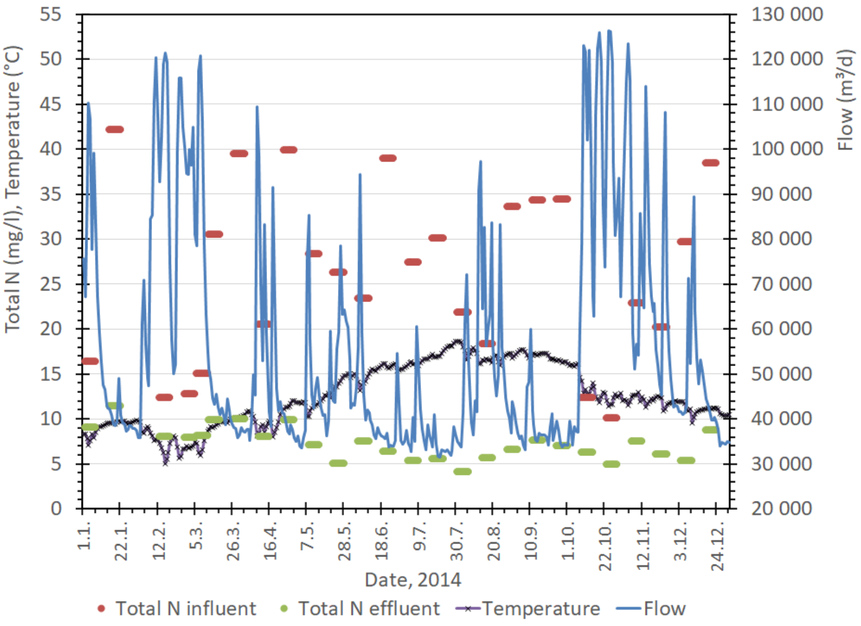


Figure 4.21. Flow through the MBBR process and average water temperature in R3 for the 4 trains at NRA for every day of the year 2014. Plant influent and effluent total N concentrations in weekly flow proportional samples are also shown.

The on-line measurements shown in Table 4.12 are strict averages over the given week and not flow proportional. However, these on-line data can be used to estimate specific nitrification rates and denitrification rates, using the procedure described in section 4.3.1. No NOx-N data were available for week 10, due to probe error. The NOx-N measured in R6 in week 4 is also questionable because it is significantly higher than the effluent total N concentration.

Calculated post-denitrification rates showed very large variations, but no pattern of temperature dependency. The average rate over all these weeks was 1.07 g NOx-N/m^2-d. It is, however, strange that the on-line data indicate a post-denitrification rate as high as 2.28 g NOx-N/m^2-d for week 8 with the lowest temperature (6.4 °C) and the highest flow (105,277 m^3/d). Pre-denitrification is expected to be very low at such a high flow and low temperature, and not surprisingly the on-line data indicate a negative pre-denitrification rate in week 8 due to the problems discussed in section 4.3.1 (not accounting for influent NOx-N and nitrification of organic N). Removing this negative value, the average pre-denitrification rate was 0.28 g NOx-N/m^2-d.

Calculated nitrification rates over R2 + R3 + R4 varied from 0.32 to 0.52 g NH_4-N/m^2-d for the weeks shown in Table 4.12. The average rate was 0.41 g NH_4-N/m^2-d. Plotting nitrification rates versus temperature, the trend line indicated a temperature dependency of $\theta = 1.08$ in the temperature range from 6.4 to 11.6 °C.

Table 4.12. Temperatures, flow rates, concentrations and nitrogen loads for NRA in the winter/spring of 2014. Also shown are on-line NH_4-N and NOx-N measurements (as weekly averages), plus calculated specific nitrification and denitrification rates based on these on-line measurements.

Parameter	**Week and date, 2014**							
	4	**8***	**10**	**11***	**12**	**14**	**16**	**18**
	16/1-22/1	**13/2-19/2**	**27/2-5/3**	**6/3-12/3**	**13/3-19/3**	**27/3-2/4**	**10/4-16/4**	**24/4-30/4**
Temperature, °C	9.6	6.4	6.9	7.2	9.2	10.2	9.1	11.6
Flow rate, m^3/d	41352	105277	95911	88552	45228	37177	64650	38062
Load:								
Total N, kg/d	1741	1295	1228	1328	1379	1468	1325	1519
Influent concentrations:								
BOD_5, mg/l	280	51	58	76	210	290	120	220
COD, mg/l	530	160	160	180	440	620	310	600
Total N, mg/l	42.1	12.3	12.8	15.0	30.5	39.5	20.5	39.9
Total P, mg/l	4.92	1.52	1.66	1.59	4.40	5.61	2.50	5.58
Effluent concentrations, mg/l								
BOD_5, mg/l	1.5	5	5	6	1.5	5	4	1.5
COD, mg/l	35	15	30	32	34	33	15	34
Total N, mg/l	11.4	7.98	7.93	8.16	9.91	9.95	8.04	9.82
Total P, mg/l	0.18	0.18	0.21	0.23	0.18	0.19	0.19	0.20
Removal efficiencies:								
BOD_5, %	99.5	60.2	91.4	92.1	99.3	98.3	96.7	99.3
COD, %	93.4	90.6	81.3	82.2	92.3	94.7	95.2	94.3
Total N, %	72.9	35.1	38.0	45.6	67.5	74.8	60.8	75.4
Total P, %	96.3	88.2	87.3	85.5	95.9	96.6	92.4	96.4
On-line measurements:								
Primary effluent NH_4-N, mg/l	31.4	8.8	10.48	13.78	27.1	34.8	16.7	30.9
R4 effluent NH_4-N, mg/l	1.07	1.50	2.17	2.26	4.41	4.45	2.10	1.57
R4 effluent NOx-N, mg/l	17.5	9.60	- - -	8.44	14.5	18.1	13.3	17.9
R6 effluent NOx-N, mg/l	***14.2***	3.42	- - -	6.85	8.49	9.04	9.09	9.43
Rates calculated from on-line data:								
Nitrification rate,g NH_4-N/m-d**	0.516	0.315	0.326	0.417	0.422	0.464	0.388	0.459
Pre-DN rate, g NOx-N/m^2-d §	0.420	-0.195	- - -	0.212	0.293	0.363	0.067	0.346
Post-DN rate,g NOx-N/m^2-d §§	***0.485***	2.28	- - -	0.491	0.954	1.18	0.951	1.12

* Water in by-pass. Given flows, loads and effluent concentrations are for water that has been through the entire treatment plant.

** Based on total biofilm surface area in R2 + R3 + R4.

§ Based on biofilm surface area in R1.

§§ Based on biofilm surface area in R5.

It can be seen from Figure 4.21 that high flow rates and low influent concentrations have a very large influence on the total N removal efficiency. In week 44 (23/10-29/10) the flow through the nitrogen removal stage was 106,262 m^3/d and only 51 % total N removal was achieved at a temperature of 12.0 °C. In week 48 (4/12-10/12), however, more than 82 % total N was removed at an average flow of 52,092 m^3/d and a temperature of 11.2 °C.

To separate the temperature effect from the effect of dilute wastewater we can compare the results from week 8 and week 44. These two weeks had similar but very high flows. Average removal was 550 kg N/d in week 44 (12.0 °C) and 455 kg N/d in week 8 (6.4 °C). This is a difference in removed total N of only 20 % and shows that the temperature effect was moderate and only 3.3 % per °C.

Because some nitrogen is removed by primary sedimentation and some is removed by assimilation the real temperature effect on respectively nitrification and denitrification is somewhat higher, but as shown in Chapter 3, increasing the DO concentration at lower temperatures can mask the temperature dependency of nitrification.

4.3.3 Results for NRA in 2016

Figure 4.22 shows flow through the MBBR process and average water temperature in R3 for the 4 trains at NRA for every day of the year. Plant influent and effluent total N concentrations are also shown, based on 25 sets of weekly flow proportional samples. Flows that by-passed the MBBR stage are not shown, but for flows above approximately 90,000 m³/d, significant amounts of water were by-passed. The lowest water temperature was 5.7 °C on February 9.

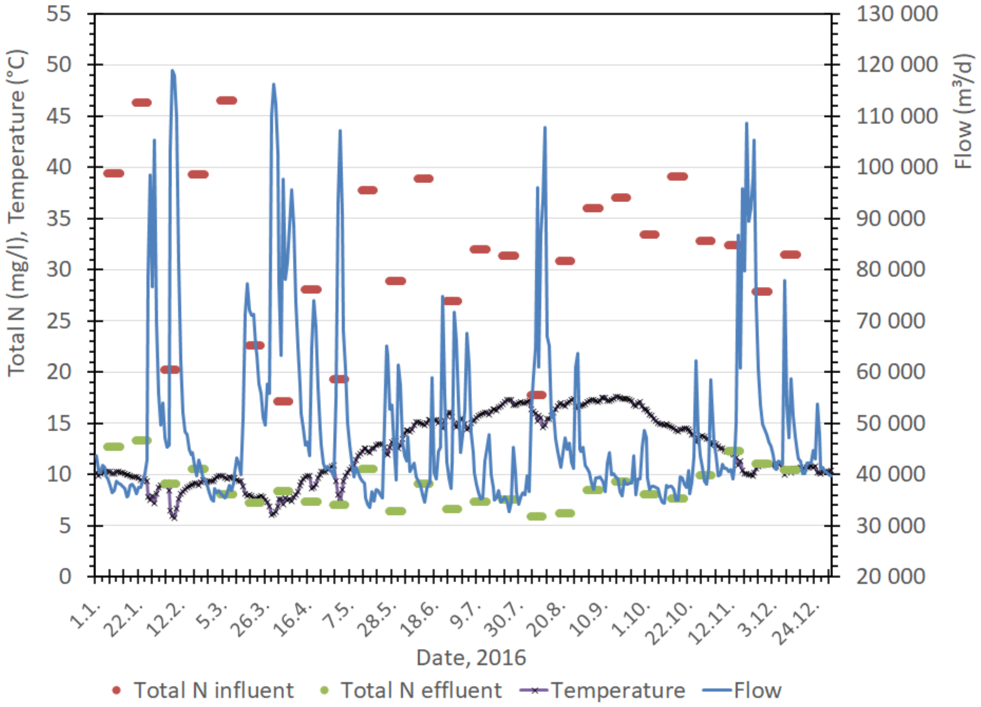


Figure 4.22. Flow through the MBBR process and average water temperature in R3 for the 4 trains at NRA for every day of the year 2016. Plant influent and effluent total N concentrations in weekly flow proportional samples are also shown.

For the part of the year with the lowest water temperatures additional results are shown in Table 4.13, as averages for the given week. The total N removal efficiencies for week 4 through week 18 were more or less inversely proportional to the influent flows. As expected at this time of the year, the highest flow rates correspond to the lowest temperatures. With about 84.000 m³/d (week 6 and week 14) the total N removal efficiency was 52 – 55 % and the average temperature in R3 was 7.4 °C. With a flow of about 37,000 m³/d (week 10) 83 % total N was removed at an average temperature of 9.7 °C.

Table 4.13. Temperatures, flow rates, concentrations and nitrogen loads for NRA in the winter/spring of 2016. Also shown are on-line NH_4-N and NOx-N measurements (as weekly averages), plus calculated specific nitrification and denitrification rates based on these on-line measurements.

Parameter	**Week and date, 2016**							
	4 **21/1-27/1**	**6*** **4/2-10/2**	**8** **18/2-24/2**	**10** **3/3-9/3**	**12** **17/3-23/3**	**14** **31/3-6/4**	**16** **14/4-20/4**	**18*** **28/4-4/5**
Temperature, °C	9.3	7.4	9.2	9.7	7.8	7.4	9.4	9.1
Flow rate, m^3/d	44536	84069	41172	36714	65163	84043	57185	73497
Load:								
Total N, kg/d	2062	1698	1618	1707	1466	1437	1601	1411
Influent concentrations:								
BOD_5, mg/l	320	120	310	270	170	91	160	120
COD, mg/l	1100	310	750	570	350	230	410	310
Total N, mg/l	46.3	20.2	39.3	46.5	22.5	17.1	28.0	19.2
Total P, mg/l	7.84	2.52	5.89	5.81	3.14	2.22	3.99	2.64
Effluent concentrations, mg/l								
BOD_5, mg/l	4	7	14	5	4	8	1.5	1.5
COD, mg/l	36	15	53	38	15	32	30	15
Total N, mg/l	13.3	9.00	10.5	7.99	7.19	8.29	7.30	6.97
Total P, mg/l	0.25	0.30	0.25	0.41	0.19	0.26	0.12	0.17
Removal efficiencies:								
BOD_5, %	98.8	94.2	95.5	98.1	97.6	91.2	99.1	98.8
COD, %	96.7	95.2	92.9	93.3	95.7	86.1	92.7	95.2
Total N, %	71.3	55.4	73.3	82.8	68.0	51.5	73.4	63.7
Total P, %	96.8	88.1	95.8	92.9	93.9	88.3	97.0	93.6
On-line measurements:								
Primary effluent NH_4-N, mg/l	25.7	13.7	27.6	28.5	12.8	11.3	20.2	15.0
R4 effluent NH_4-N, mg/l	2.19	1.32	1.41	2.43	1.78	3.05	2.68	0.77
R4 effluent NOx-N, mg/l	13.4	10.0	13.5	15.1	8.80	8.48	11.6	10.6
R6 effluent NOx-N, mg/l	8.03	6.55	6.07	4.60	4.74	5.73	5.15	6.98
Rates calculated from on-line data:								
Nitrification rate, g NH_4-N/m^{2}d**	0.431	0.428	0.443	0.393	0.296	0.286	0.412	0.429
Pre-DN rate, g NOx-N/m^{2}d [§]	0.358	0.156	0.414	0.321	0.116	-0.013	0.271	0.212
Post-DN rate, g NOx-N/m^2-d [§§]	0.841	1.03	1.08	1.34	0.926	0.807	1.28	0.921

* Water in by-pass. Given flows, loads and effluent concentrations are for water that has been through the entire treatment plant.

** Based on total biofilm surface area in R2 + R3 + R4.

[§] Based on biofilm surface area in R1.

[§§] Based on biofilm surface area in R5.

Week 6 and week 18 had almost identical influent concentrations of BOD_5, COD, total N and total P, but the flow was 14 % higher in week 6 than in week 18. Removal was 941 kg N/d in week 6 at 7.4°C, and 899 kg N/d in week 18 at 9.1 °C. Again, this indicates that the temperature was not the most important factor for the nitrogen removal in this temperature range.

The on-line measurements shown in Table 4.13 are strict averages over the given week and not flow proportional. However, these on-line data can be used to estimate specific nitrification rates and denitrification rates, using the procedure described in section 4.3.1.

Calculated post-denitrification rates showed variations from 0.81 to 1.34 g NOx-N/m²-d, with an average over all these weeks of 1.03 g NOx-N/m²-d. Plotting post-denitrification rates versus temperature, the trend line indicated a temperature dependency of θ = 1.11 in the very narrow temperature range from 7.4 to 9.7 °C. Pre-denitrification again showed very low rates at high flows, and negative values in week 14, most likely because of the same reasons as previously mentioned in section 4.3.1 (not accounting for influent NOx-N and nitrification of organic N). Removing the negative value in week 14, the average pre-denitrification rate was 0.26 g NOx-N/m²-d.

Calculated nitrification rates over R2 + R3 + R4 varied from 0.29 to 0.44 g NH_4-N/m²-d for the weeks shown in Table 4.13. The average rate was 0.39 g NH_4-N/m²-d. Plotting nitrification rates versus temperature, the trend line indicated a temperature dependency of θ = 1.12 in the very narrow temperature range from 7.4 to 9.7 °C. However, if we delete the result from week 14 (with negative pre-denitrification rate), the trend line for nitrification shows a temperature dependency of only θ = 1.07.

The results from week 32 (4/8-10/8, see Figure 4.22) are yet another example of influent concentrations and flow rates having the largest influence on the nitrogen removal. The flow in week 32 was about the same as in week 18, but the removal of total N was slightly less than 67 % even though the average temperature was as high as 15.6 °C.

4.3.4 Results for NRA in 2018

Figure 4.23 shows flow through the MBBR process and average water temperature in R3 for the 4 trains at NRA for every day of the year. Plant influent and effluent total N concentrations are also shown, based on 25 sets of weekly flow proportional samples. Flows that by-passed the MBBR stage are not shown, but for flows above approximately 85,000 m³/d, significant amounts of water were by-passed. The lowest water temperature was 4.7 °C on April 19.

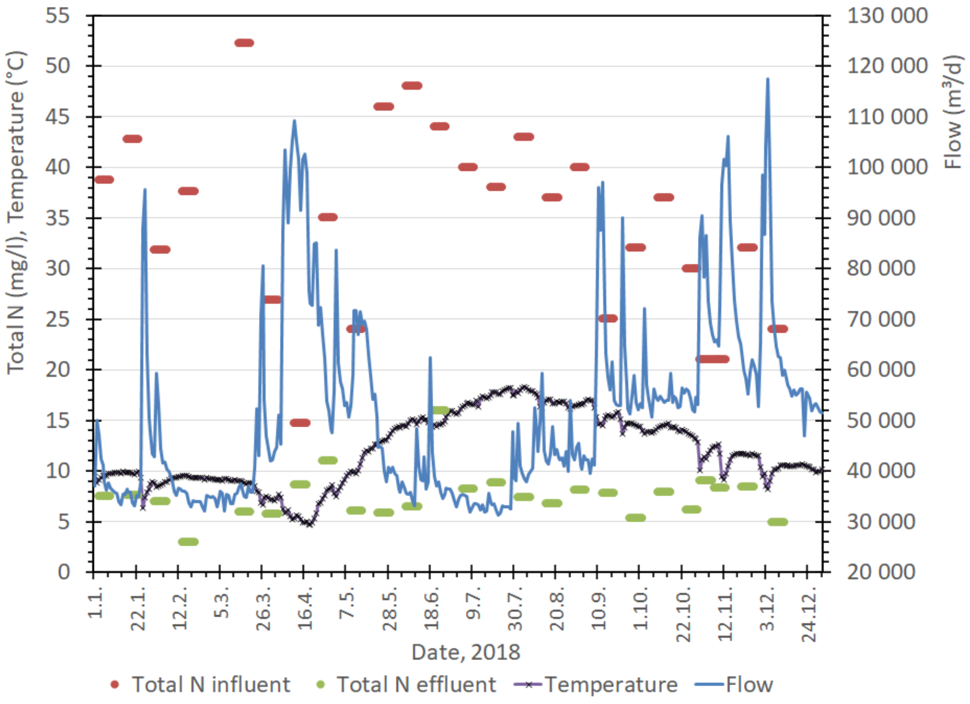


Figure 4.23. Flow through the MBBR process and average water temperature in R3 for the 4 trains at NRA for every day of the year 2018. Plant influent and effluent total N concentrations in weekly flow proportional samples are also shown.

For the part of the year with the lowest water temperatures additional results are shown in Table 4.14, as averages for the given week. This will typically be in connection with melting snow and precipitation in the spring. Week 12 had low water flow, high influent concentrations and high removal efficiencies. Removal of total N was 89 % at an average temperature in R3 of 8.8 °C. In week 14, with approximately half the influent concentration and a bit higher flow, the removal of total N dropped to 79 % at a temperature of 7.3 °C. Week 16 had extremely high flow, very low influent concentrations and an average temperature of only 5.2 °C. The removal efficiency for total N then dropped all the way down to 41 %. In week 18, with high influent total N and low influent BOD$_5$ concentrations the removal of total N increased again, up to 69 % at an average temperature of 8.0 °C.

Table 4.14. Temperatures, flow rates, concentrations and nitrogen loads for NRA in the spring of 2018. Also shown are on-line NH$_4$-N and NOx-N measurements (as weekly averages), plus calculated specific nitrification and denitrification rates based on these on-line measurements.

Parameter	Week and date, 2018				
	12	**14**	**16***	**18**	**20**
	14/3-20/3	**28/3-3/4**	**11/4-17/4**	**25/4-1/5**	**9/5-15/5**
Temperature, °C	8.9	7.3	5.2	8.0	10.2
Flow rate, m^3/d	36185	45023	101513	55201	66149
Load:					
Total N, kg/d	1889	1211	1492	1932	1588
Influent concentrations:					
BOD$_5$, mg/l	470	200	68	100	120
COD, mg/l	910	490	180	430	290
Total N, mg/l	52.2	26.9	14.7	35.0	24.0
Total P, mg/l	9.38	4.36	2.29	3.73	3.00
Effluent concentrations, mg/l					
BOD$_5$, mg/l	1.5	4	11	5	4
COD, mg/l	15	15	34	33	15
Total N, mg/l	5.90	5.75	8.61	11.0	6.10
Total P, mg/l	0.18	0.14	0.31	0.18	0.12
Removal efficiencies:					
BOD$_5$, %	99.7	98.0	83.8	95.0	96.7
COD, %	98.4	96.9	81.1	92.3	94.8
Total N, %	88.7	78.6	41.4	68.6	74.6
Total P, %	98.1	96.8	86.5	95.2	96.0
On-line measurements:					
Primary effluent NH$_4$-N, mg/l	30.9	18.3	8.19	19.2	15.1
R4 effluent NH$_4$-N, mg/l	3.38	1.62	1.35	1.82	0.83
R4 effluent NOx-N, mg/l	14.3	11.5	7.32	10.0	9.58
R6 effluent NOx-N, mg/l	4.70	4.89	5.27	5.40	4.70
Rates calculated from on-line data:					
Nitrification rate, g NH$_4$-N/m^{2}d **	0.410	0.309	0.286	0.395	0.389
Pre-DN rate, g NOx-N/m^2-d §	0.381	0.186	-0.039	0.324	0.248
Post-DN rate, g NOx-N/m^2-d §§	1.22	1.04	0.728	0.889	1.13

* Water in by-pass. Given flows, loads and effluent concentrations are for water that has been through the entire treatment plant.

** Based on total biofilm surface area in R2 + R3 + R4.

§ Based on biofilm surface area in R1.

§§ Based on biofilm surface area in R5.

The highest amount of nitrogen removed was in week 12, with 1,675 kg N/d, and in week 18, with 1,325 kg N/d. In week 16, with 5.2 °C in R3, only 618 kg N/d was removed. In addition to low influent concentrations and very high flow, the effluent total P concentration indicates that the effluent contained more TSS than normal, which may have contributed to low removal of total N in week 16.

The on-line measurements shown in Table 4.14 are strict averages over the given week and not flow proportional. However, these on-line data can be used to estimate specific nitrification rates and denitrification rates, using the procedure described in section 4.3.1.

Calculated post-denitrification rates showed variations from 0.73 to 1.22 g NOx-N/m^2-d, with an average over all these weeks of 1.00 g NOx-N/m^2-d. Plotting post-denitrification rates versus temperature, the trend line indicated a temperature dependency of $\theta = 1.12$ in the temperature range from 5.2 to 10.2 °C. Pre-denitrification again showed low rates at high flows, and negative value in week 16 with extremely high flow. This negative value was most likely caused by the same factors previously mentioned in section 4.3.1 (not accounting for influent NOx-N and nitrification of organic N). Removing the negative value in week 16, the average pre-denitrification rate was 0.29 g NOx-N/m^2-d. With the negative value removed the trend line indicated a temperature dependency for pre-denitrification of $\theta = 1.09$ in the temperature range from 7.3 to 10.2 °C. However, we know from multiple pilot-tests that the most important factors for pre-denitrification are the availability of easily biodegradable carbon and limited input of oxygen to the reactor, and these factors can easily mask the true temperature effect.

Calculated nitrification rates over R2 + R3 + R4 varied from 0.29 to 0.41 g NH_4-N/m^2-d for the weeks shown in Table 4.14. The average rate was 0.36 g NH_4-N/m^2-d. Plotting nitrification rates versus temperature, the trend line indicated a temperature dependency of $\theta = 1.08$ in the temperature range from 5.2 to 10.2 °C. However, if we delete the result from week 16 (with negative pre-denitrification rate), the trend line for nitrification shows a temperature dependency of $\theta = 1.07$ in the temperature range from 7.3 to 10.2 °C.

4.4 Gardermoen WWTP

The Gardermoen WWTP (GRA) was started up in 1998, at the same time as the new Oslo airport was opened. GRA is located just outside the airport fence and serves the airport plus the Ullensaker and Nannestad municipalities. A simplified flowsheet of the liquid processes in the main plant at GRA is shown in Figure 4.24, and key data for the main plant are shown in Table 4.15.

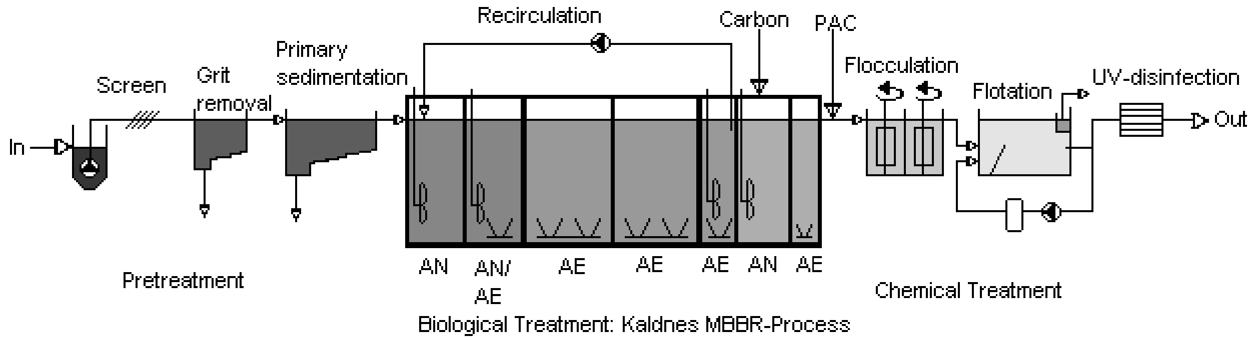


Figure 4.24. Simplified flowsheet of the liquid treatment processes in the main plant at the Gardermoen WWTP.

Originally the most dilute runoff of airport de-icing chemicals from the de-icing platforms and along the taxi ways and runways was treated in on-site soil infiltration systems. However, these systems rapidly failed, and the dilute runoff had to be rerouted to GRA. A biological pre-treatment facility, consisting of two MBBRs in parallel, was therefore built next to the main plant at GRA and started up in the winter of 2004. This pre-treatment plant gets the most dilute aircraft deicing fluid runoff pumped directly from storage lagoons at the airport, plus necessary nutrients in the form of some of the primary effluent from the main treatment plant. One of the DAF trains in the main plant has been designated for removal of particulate material from the pre-treatment MBBR effluent, before this effluent is returned to the inlet of the biological stage in the main plant.

The slightly more concentrated fraction of collected aircraft de-icing fluid (monopropylene glycol, MPG) from the de-icing platform is collected in separate lagoons at the airport and pumped to GRA for use as external carbon source for post-denitrification. This is internally called B-glycol and has a COD in the range of 10 to 20 g/l. The most concentrated fraction of the collected de-icing chemical runoff is called A-glycol and was originally put on tanker trucks and used as external carbon source at the Nordre Follo and NRA WWTPs. However, this contract was terminated a few years ago. Collected A-glycol is now recycled and comes back to market in different products, one example being windshield washer fluid.

Table 4.15. Key data for the liquid treatment processes in the main plant at the Gardermoen WWTP in 2019 and 2020.

Aerated sand and fat traps	Total volume:	150 m^{3}	2 trains
Primary sedimentation	Total surface area: Water depth:	420 m^{2} 3.3 m	2 trains
MBBRs	Total wet volume: Water depth:	5790 m^{3} 6.5 m	2 trains
Flocculation	Total volume:	180 m^{3}	2 reactors in series 3 trains
Flotation	Total surface area: Water depth:	215 m^{2} 2.5 m	3 trains *
Disinfection	UV-lamps		2 banks

Details for the MBBR process:

Reactor	Wet volume	Operation mode (AN = anoxic AE = aerobic)	Filling of K1	Total biofilm surface area (2 trains)
R1	2 x 420 m^{3}	AN	50 %	210,000 m^{2}
R2	2 x 420 m^{3}	AN or AE	50 %	210,000 m^{2}
R3	2 x 695 m^{3}	AE	60 %	417,000 m^{2}
R4	2 x 695 m^{3}	AE	60 %	417,000 m^{2}
R5	2 x 180 m^{3}	AE (de-Ox)	40 %	72,000 m^{2}
R6	2 x 375 m^{3}	AN	50 %	187,500 m^{2}
R7	2 x 110 m^{3}	AE	50 %	55,000 m^{2}

* One train used for solids separation after biological pre-treatment of dilute deicing fluid runoff from the Oslo airport.

Surface runoff from the Oslo airport also includes formate from runway de-icing. This runoff is very cold (close to 0 °C), is temporarily stored in lagoons at the airport, and then pumped to the main plant at GRA. A heat exchanger was installed a few years ago, using the effluent from GRA to pre-heat the cold surface runoff from the airport. This has significantly increased the temperature in the treatment plant.

The original filling of biofilm carriers was less than what the plant was designed for. In the fall of 2017 more K1 biofilm carriers were added to the MBBRs in the main plant at GRA, resulting in biofilm carrier fillings and total biofilm surface areas shown in Table 4.15, which are still lower than the original design.

GRA was originally designed for the flows and loads shown in Table 4.16. The consent values were changed in 2013, to be more in line with the EU-directive and the requirements at other Norwegian treatment plants. The requirement for nitrogen removal stayed the same, but the requirements for removal of organic matter and phosphorus were considerably more lenient in the 2013 revision than what was originally required in 1998.

Table 4.16. Original design basis for the liquid treatment processes at GRA. Loads are for 80-percentiles.

	Flows, loads and temperature	**Original consent values, 1998 (annual average)**	**Revised consent values, 2013**
$Q_{average}$	16,000 m^3/d	- - -	- - -
Q_{design}	920 m^3/h	- - -	- - -
$Q_{maxdesign}$	1,300 m^3/h	- - -	- - -
Total COD	5,190 kg/d	- - -	≥ 75 % removal, or ≤ 125 mg/l
Filtered COD	2,075 kg/d	- - -	- - -
Total BOD_5	2,180 kg/d	≤ 10 mg/l	≥ 70 % removal, or ≤ 25 mg/l
TSS	3,115 kg/d	- - -	- - -
Total N	604 kg/d	≥ 70 % removal	≥ 70 % removal (annual average)
Total P	96 kg/d	≤ 0.2 mg/l	≥ 90 % removal (annual average)
Temperature range (monthly average)	5 – 14 °C	- - -	- - -

Going through the flowsheet in Figure 4.24 GRA has screens (3 – 6 mm), aerated sand and fat traps, and MBBR with 7 reactors in series. R1 is always anoxic for pre-denitrification. R2 can be either anoxic or aerobic. R3 + R4 are always aerobic and aerated. R5 is aerobic, but normally with just propeller mixing to reduce the DO before nitrified water is returned back to R1 for pre-denitrification or passed on to R6 for post-denitrification. Used de-icing fluid (MPG) is added to R6 as an external carbon source. R7 is a small post aeration reactor. For removal of particles and precipitation of phosphorus pre-polymerized aluminum chloride (PAC, type Ekoflock) is added, together with an anionic polymer (Zetag 4105) as a flocculation aid. After flocculation and dissolved air flotation (DAF), a UV-disinfection system is in use from May through October. This is to assure bathing water quality in the recipient.

GRA has fairly standard sludge treatment, with two anaerobic digestors in series. The first digestor is thermophilic (55 °C) to ensure that the sludge is hygienized, and the second digestor is mesophilic (37 °C)

4.4.1 Overall results for Gardermoen WWTP in 2019 and 2020.

Tables 4.17 and 4.18. show a summary of results for GRA in 2019 and 2020, respectively. Treatment efficiencies were significantly better than required in both years, with average removal of about 85 % total N, 96 % total P, 96-97 % COD, 99 % BOD_5 and 97 % TSS. Effluent NH_4-N concentrations were most of the time very low, and hence limiting with respect to nitrification rates.

Overall, the results were excellent for a plant that in 2019 received a load of total N and organic matter that was 1.5 to 1.8 times higher than the design load. In 2020 loads were lower, but still well above the design loads. Especially the total N loads were significantly lower than in 2019. The reason for this was the Covid 19 pandemic and the reduced activity at the Oslo airport for the last 8-9 months of the year. In a normal year, more than 20 % of the total N to the treatment plant originates from the airport.

Table 4.17. Summary of results for the Gardermoen WWTP in 2019.

Parameter	Average	Median	Min.	80-percentile	Max.	N
Flow, m^3/d	12,110	11,931	7,919	13,689	21,804	358
Temp. in R4, °C	11.2	10.6	6.7	14.2	16.5	358
Loads:						
BOD_5, kg/d	3,008	3,078	1,882	3,324	3,882	24
Total COD, kg/d	8,590	8,540	5,315	9,206	13,123	204
Filtered COD, kg/d	2,507	2,335	1,807	3,058	3,702	32
TSS, kg/d	4,561	4,500	3,444	4,937	6,774	32
Total N, kg/d	871	851	665	928	1,394	204
Total P, kg/d	101	100	77	108	134	205
Influent concentrations:						
BOD_5, mg/l	253	245	110	320	360	24
Total COD, mg/l	713	711	368	822	1,262	208
Filtered COD, mg/l	202	193	138	228	297	32
TSS, mg/l	370	355	273	415	629	32
Total N, mg/l	73	70	40	84	130	208
Total P, mg/l	8.4	8.3	4.0	9.6	12.2	209
Effluent concentrations:						
BOD_5, mg/l	2.7	1.5	1.5	3.6	9.8	24
Total COD, mg/l	24	22	11	28	101	208
Filtered COD, mg/l	15	14	5	18	29	36
TSS, mg/l	9.7	9.0	1.0	12	28	32
Total N, mg/l	11.2	10.9	3.7	14.1	28.0	208
NH_4-N, mg/l	3.6	2.0	0.1	6.0	16.3	31
NO_3-N, mg/l	6.3	5.5	3.4	8.2	14.9	32
Total P, mg/l	0.37	0.32	0.14	0.44	2.01	209
Total P on filtered sample, mg/l	0.04	0.03	0.03	0.03	0.15	208

Flows through the MBBR process and water temperatures in R4 for GRA for every day of the year 2019 are shown in Figure 4.25. Also shown are plant influent and effluent total N concentrations in 24 h flow proportional samples. The lowest temperatures were observed from March 29 to April 3, with 6.7 to 7.7 °C in R4, when the flow increased by about 60 % compared to January and February. During this week with low temperatures removal of total N varied from 78.2 to 86.4 %, with influent concentrations from 52 to 59 mg total N/l and effluent concentrations from 7.1 to 11.8 mg total N/l.

Table 4.18. Summary of results for the Gardermoen WWTP in 2020.

Parameter	Average	Median	Min.	80-percentile	Max.	N
Flow, m³/d	11,843	11,073	7,383	14,404	22,053	365
Temp. in R4, °C	10.7	10.6	5.4	13.3	18.4	365
Loads:						
BOD$_5$, kg/d	2,600	2,434	1,679	3,149	3,812	24
Total COD, kg/d	7,545	7,519	5,032	8,402	12,165	212
Filtered COD, kg/d	2,146	2,144	1,360	2,585	2,944	43
TSS, kg/d	3,850	3,929	2,835	4,345	5,144	43
Total N, kg/d	685	679	528	748	868	212
Total P, kg/d	89	89	66	97	126	212
Influent concentrations:						
BOD$_5$, mg/l	229	226	118	264	372	24
Total COD, mg/l	666	654	341	770	1,187	213
Filtered COD, mg/l	183	181	102	212	282	43
TSS, mg/l	342	334	211	411	533	43
Total N, mg/l	61	60	30	70	82	213
Total P, mg/l	7.9	7.9	4.2	9.2	13.7	213
Effluent concentrations:						
BOD$_5$, mg/l	2.3	1.0	1.0	4.0	6.0	24
Total COD, mg/l	29	27	17	33	95	213
Filtered COD, mg/l	20	18	10	23	71	43
TSS, mg/l	9.1	8.0	1.0	14	22	43
Total N, mg/l	9.2	7.9	3.4	10.5	42	211
NH$_4$-N, mg/l	1.8	0.3	0.1	2.5	26.2	43
NO$_3$-N, mg/l	5.2	4.8	1.6	7.1	16.5	43
Total P, mg/l	0.31	0.26	0.12	0.40	1.3	213
Total P on filtered sample, mg/l	0.04	0.03	0.03	0.03	0.42	214

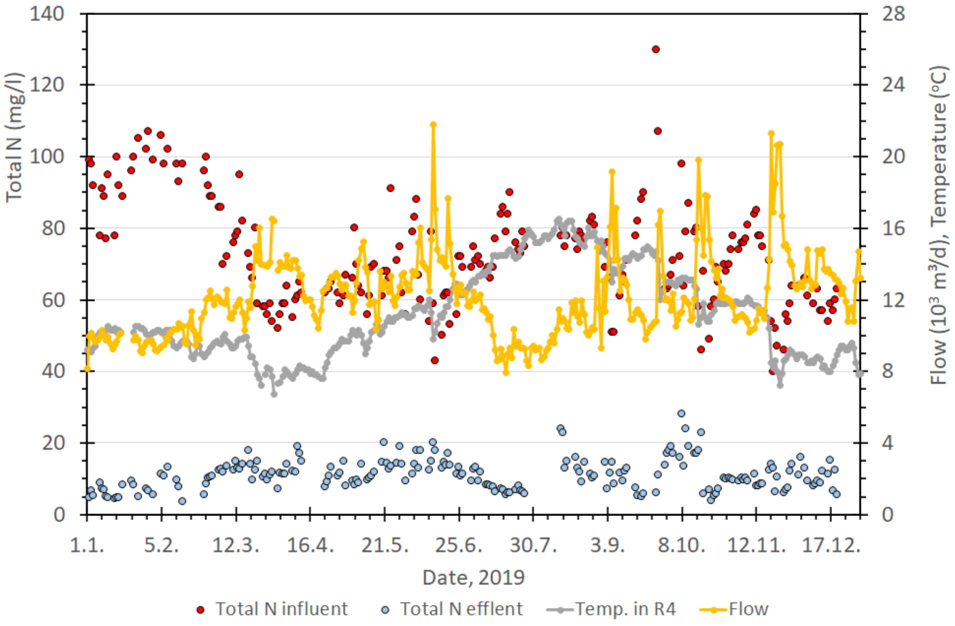


Figure 4.25. Flow through the MBBR process and water temperature in R4 for GRA for every day of the year 2019. Plant influent and effluent total N concentrations in 24 h flow proportional samples are also shown.

On April 3 ammonium and nitrate were also measured in the effluent, showing 6.0 mg NH_4-N/l and 3.7 mg NO_3-N/l, together with 11.4 mg total N/l. An interesting observation is that the highest effluent concentrations of total N happened in August at 16 °C and October at 13 °C, so obviously other factors than low temperatures were responsible for the elevated effluent concentrations during these periods.

Flows through the MBBR process and water temperatures in R4 for GRA for every day of the year 2020 are shown in Figure 4.26. Also shown are plant influent and effluent total N concentrations in 24 h flow proportional samples. The lowest temperature on a day with complete sampling was on March 11, with 6.0 °C in R4. The flow was then about twice the dry weather flow observed in the summer. Influent total N was 41 mg N/l, while effluent concentrations were 5.8 mg NH_4-N/l, 1.9 mg NO_3-N/l and 9.3 mg total N/l. This corresponded to 77.3 % removal of total N. Again, the highest effluent total N concentrations were observed in early July and late September, at significantly higher temperatures. However, the high effluent concentrations in September can be explained by train 1 being shut down for maintenance.

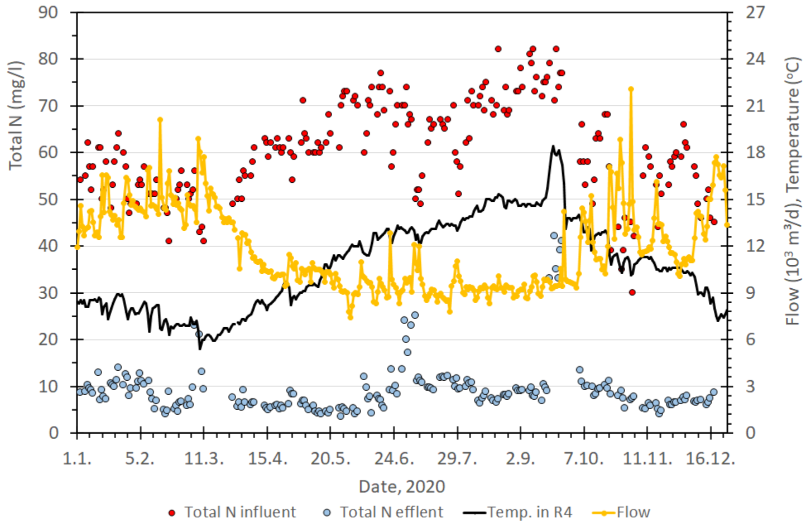


Figure 4.26. Flow through the MBBR process and water temperature in R4 for GRA for every day of the year 2020. Plant influent and effluent total N concentrations in 24 h flow proportional samples are also shown.

4.4.2 Specific reaction rates

4.4.2.1 Basis for calculations

GRA has data for flows, temperature in R4, DO in R1, R4 and R5, total COD and filtered COD in to biological stage and in final effluent, total N in to biological stage and in final effluent, NH_4-N in to biological stage, out of R5 in train 2 and in final effluent, NO_2-N + NO_3-N out of R5 in train 2 and in final effluent, plus added amount of MPG as external carbon source.

For calculating the specific nitrification and denitrification rates the following assumptions have been made:

- Nitrogen assimilated due to influent organic matter to biological stage set at 3.6 % of applied filtered COD.
- Assimilation of nitrogen due to addition of external carbon source to R6 calculated based on a sludge yield of 0.253 g VS/g COD at 15 °C and a temperature correction factor of θ=1.064 (Rusten *et al.*, 1996). Sludge yield increases at lower temperatures. Sludge produced from external carbon source assumed to have a nitrogen content of 5.0 % of VS.

- Amount of nitrogen removed by pre-denitrification based on total N in to biological stage, minus total N in plant effluent, minus N removed by assimilation, and minus N removed by post-denitrification.
- Amount of nitrogen removed by post-denitrification based on NOx-N (NO_2-N + NO_3-N) out of R5 in train 2, NOx-N in final effluent, and water flow through train 2. It is assumed that train 1 and train 2 had equal post-denitrification rates.
- Nitrified amount is calculated for train 2 based on flow through train 2, NH_4-N in to biological stage and NH_4-N out of R5 in train 2. Assimilated nitrogen due to influent organic matter to the biological stage has not been subtracted, because some particulate and organic N will be hydrolyzed in both anoxic and aerobic reactors and thus increase the NH_4-N load. This is expected to give a conservative estimate of the nitrified amount, since total N in to the biological stage was on average 38 % higher than NH_4-N, giving an average amount of more than 210 kg/d of particulate and organic N compared to only 88 kg/d assimilated N.
- Calculating the biofilm surface area that contributes to nitrification is based on multiple assumptions. Initially the organic load on the first aerobic reactor is calculated, based on the amount of COD in to the biological stage, minus the amount of organic matter consumed by pre-denitrification. Secondly, it is assumed that nitrification can begin when the organic load on the aerobic reactor is ≤ 10 g COD/m²-d at 15 °C, using a temperature coefficient of $\theta=1.08$. For consumption of organic matter in an aerobic reactor a reaction rate of 20 g COD/m²-d at 15 °C has been used, with the same temperature correction as above. If the organic load on an aerobic reactor is higher than the maximum organic load for the start of nitrification at the given temperature, no nitrification is assumed to take place in this reactor. If the organic load is lower, then the biofilm surface area that contributes to nitrification in this reactor is calculated based on the difference between the maximum organic load and the actual (calculated) organic load, divided by the maximum organic load and multiplied by the total biofilm surface area in the reactor.

4.4.2.2 Pre-denitrification rates

Pre-denitrification rates are shown in Figure 4.27, for the period from August 10, 2019, to June 5, 2020. Also shown are corresponding temperatures in R4, and C/N ratios in to the biological stage based on both total COD and filtered COD.

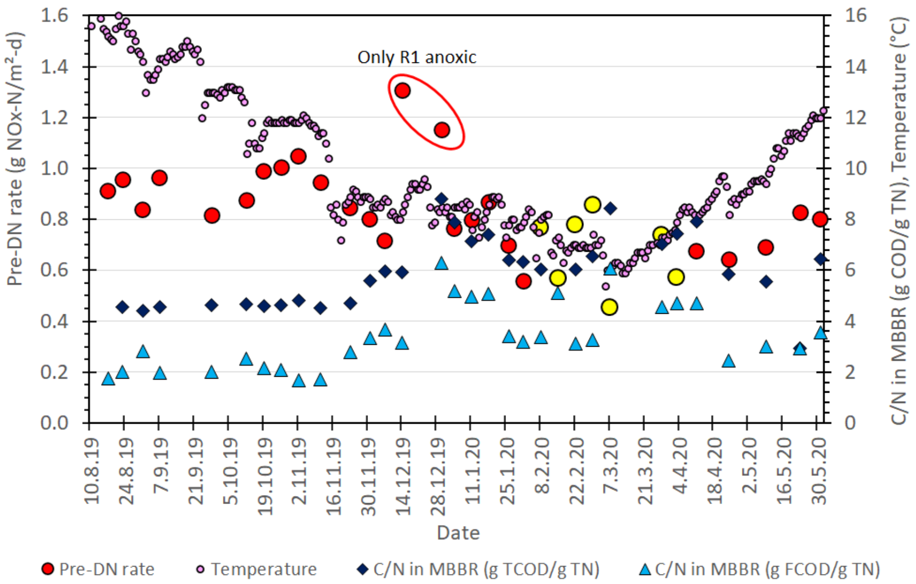


Figure 4.27. Pre-denitrification rates in MBBRs at GRA. Normally R1 and R2 in both trains were anoxic. For the two data points marked, only R1 was anoxic. Rates shown with yellow circles are from a period with R1 and R2 anoxic in train 1, and only R1 anoxic in train 2.

Most of the time R1 and R2 were anoxic in both trains. However, in December 2019 and January 2020 there were two data points where only R1 was anoxic and these two data points are marked in Figure 4.27. Furthermore, from February 6 to April 10, 2020, only R1 was anoxic in train 2, while both R1 and R2 were anoxic in train 1. These data points are shown with yellow circles.

C/N ratios in to the biological stage were always low, but slightly higher than average around Christmas and the beginning of the new year. A lot of factors influence the pre-denitrification rates, so it is difficult to single out the temperature effect. However, the data indicate reduced rates when the temperature dropped below 8 °C. In the temperature range from 8.0 to 16.5 °C the average pre-denitrification rate was 0.88 g NOx-N/m²-d. This dropped to an average of 0.68 g NOx-N/m²-d in the temperature range from 5.4 to 8.0 °C. At both the higher and the lower temperatures, these pre-denitrification rates were higher than what was expected based on some measurements made 4 years ago, but agrees well with pre-denitrification rates measured at 7.5 – 8.0 °C during the guarantee testing of the plant (Rusten and Ødegaard, 2007).

4.4.2.3 Post-denitrification rates

Post-denitrification rates are shown in Figure 4.28, together with corresponding temperatures and C/N-ratios. No temperature dependency can be found. Some of the highest post-denitrification rates were in fact seen at the very lowest temperatures. Most probably this was due to the operators increasing the C/N ratio to make up for the slightly reduced pre-denitrification rates at the lowest temperatures. At 7 °C the post-denitrification rates were about 1.4 g NOx-N/m²-d, which is the maximum you can expect with MPG as external carbon source. The average post-denitrification rate for all the data points in Figure 4.28 was 1.2 g NOx-N/m²-d.

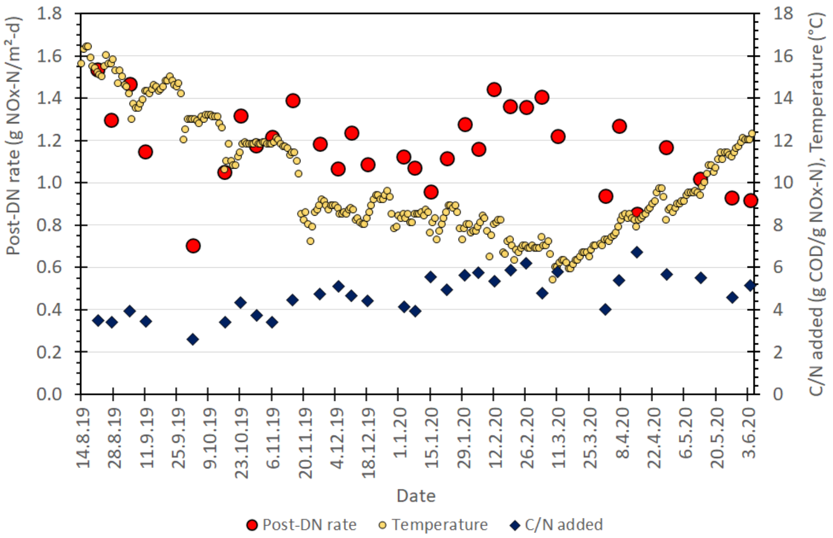


Figure 4.28. Post-denitrification rates in R6, train 2, at GRA.

4.4.2.4 Nitrification rates

Specific nitrification rates in MBBRs in train 2 are shown in Figure 4.29, together with corresponding temperatures, DO concentrations in R4 and R5, and NH_4-N concentrations in R5. When there is a high enough NH_4-N concentration in the reactor, the DO concentration can make a large difference with respect to the nitrification rate.

In two relatively short periods (second half of December 2019 and into first week of January 2020, plus from February 2 to April 10, 2020) R2 in train 2 was operated in aerobic mode. Data points with nitrification rates from these two periods are shown with yellow circles in Figure 4.29. Shortly after switching R2 from anoxic to aerobic operation, a reduced overall nitrification rate should be expected because the biofilm will need time to adjust and grow more nitrifiers in the downstream aerobic reactors that will now get a significantly lower load of organic matter. These two periods coincided with the lowest temperatures. Average specific nitrification rate with R2 in aerobic mode was 0.77 g NH_4-N/m²-d, and it was 0.99 g NH_4-N/m²-d with R2 in anoxic mode.

Specific nitrification rates versus temperature are shown in Figure 4.30. Only the days when the rates were not limited by low NH_4-N concentrations were used. This was determined by the equations developed for nitrification rates in MBBRs by Rusten *et al.* (1995). There is large scatter in Figure 4.30, since a lot of factors influence the nitrification rate. However, the trend line indicates a very low temperature dependency with an apparent θ of only 1.015 over the temperature range from 6 to 16 °C. As mentioned previously, increased DO at low temperatures can mask the true temperature dependency. A couple of data-points in May 2020 had lower DO than typically seen in February and March, and could probably have produced higher reaction rates at higher DO levels.

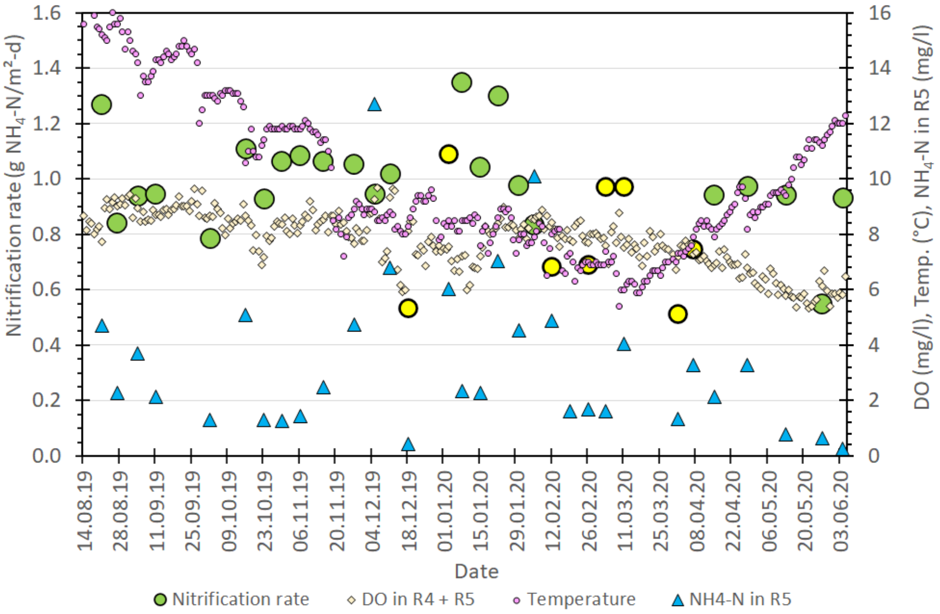


Figure 4.29. Nitrification rates in MBBRs in train 2 at GRA. Rates shown with yellow circles are from a period with only R1 anoxic in train 2. Many of the rates were limited by low NH_4-N concentrations.

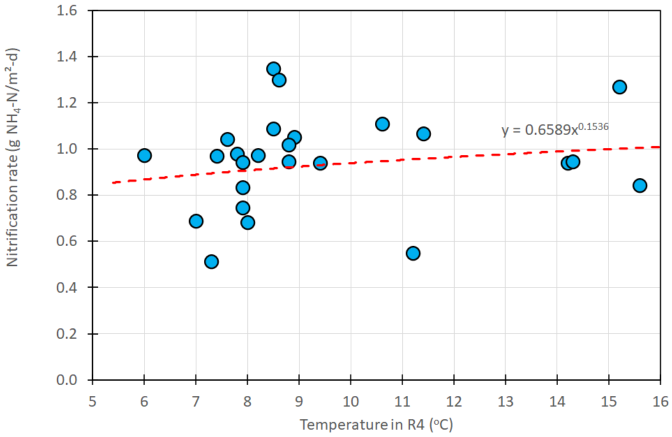


Figure 4.30. Nitrification rates in MBBRs in train 2 at GRA versus temperature in R4. Rates shown only for days when nitrification was not limited by low NH_4-N concentrations.

5 Summary and conclusions

5.1 About the report

A compilation of experiences on the influence of cold water on nitrogen removal in the moving bed biofilm reactor (MBBR) processes in publications reporting from various experiments has been carried out, as well as a comprehensive survey of operational data from 4 full-scale Norwegian nutrient removal plants based on MBBR.

The MBBR was invented and developed in Norway, a country that often experience cold wastewater, especially during snow-melt in winter and spring, and hence the majority of studies have been carried out in Norway. The Norwegian experiences are very relevant for the actual “Cold N” project.

In chapter 2 of this report a discussion on the challenges of nitrogen removal in cold wastewater is presented and the benefits for the combined pre- and post-denitrification MBBR process has been discussed.

5.2 The literature review

In chapter 3 studies from different countries (Norway, USA, Canada, Italy, and Sweden) are presented and discussed. The first studies at NTNU (Hem et al, 1994) and those carried out as part of the Norwegian N-removal program (FAN) (Rusten et al 1995b, Rusten et al 1995c) constitute the basis for the knowledge about the MBBR-processes and their design.

It was established that design should be based on area loading rates ($g/m^2_{biofilmarea} \cdot d$), that organic matter had to be removed prior to nitrification and that nitrification was linearly influenced by the DO concentration at ammonium concentrations above ca. 2 mg NH_4-N/l, and hence in most situations in practice. It was also established that the maximum nitrification rate was around 1 g NH_4-N/m^2d (at 10°C) and that the reaction order with respect to ammonium at concentrations lower than around 2 mg NH_4-N/l, was ca. 0.7. Based on this, a design model for nitrification was established (see chapter 3). It was demonstrated that the pretreatment of the water influenced the nitrification rate, not only because various pre-treatments removed organics matter, in general, to various extents, but also because good pre-treatment (such as pre-precipitation) removed colloidal, organic particles better than pre-settling for instance.

Temperature dependency in biological processes is normally expressed by the Arrhenius equation:

$\mathbf{k_2 = k_1 \cdot \theta^{(T_2 - T_1)}}$, where k_1 and k_2 are the rates at temperatures T_1 and T_2 (°C), respectively

In the literature one may find Arrhenius correction coefficients for nitrification typically around Θ = 1.10. The solubility of oxygen in water is also temperature dependent, however – the lower the temperature, the higher the DO. And hence the apparent temperature dependency in MBBR’s is lower than the actual temperature dependency. Several of the studies found that the influence of temperature, as such, was far less than expected.

In most municipal wastewater treatment plants the temperature seldom falls under 5°C, but it is demonstrated in experiments with nitrification in effluents from lagoons in Canada, that the nitrification takes place even at as low as 1 °C, at a rate that is lower, but still significant - in the order of half the rate at 20°C – equivalent to a θ–value of less than 1.05. Nitrification seems to be more easily adaptable to low temperatures than we think. However, the challenge is not so much the low temperature as such, as it is the rapid change in temperature and the corresponding diluted water at snow-melt situations. In pre-denitrification systems, therefore, there is not much to gain by increasing the recycle to above r = 2 ($r = Q_r/Q_{in}$ and hence post-denitrification was found to be very useful in order to achieve a high treatment result in cold, diluted wastewaters.

It was demonstrated in these early studies, that the pre-denitrification rate was totally dependent on the availability of easily biodegradable organic matter and that the maximum post-denitrification rate (around 2 g NO_x-N/m^2d) was 3-4 times higher than the maximum pre-denitrification rate (around 0.5 g NO_x-N/m^2d at sufficient organic matter present). A C/N-ratio > 3 g COD/g NO_x-N was found to be needed in order to reach good denitrification with ca. 4 g COD/g NO_x-N as the optimal.

Tests of the efficiency of different external carbon sources at various temperatures, demonstrated that the use of methanol and mono propylene glycol (MPG) gave similar denitrification rates (around 1.5 g NO_x-N/m^2d at 10°C) while use of ethanol doubled the denitrification rates. All three external carbon sources showed a temperature coefficient of $\theta = 1.07$. The necessary C/N-ratio increased with an average of 45 % when the temperature was reduced from 15 to 5 °C.

The benefit of having a post-denitrification stage led to the proposal, when the first full-scale N-removal plant was planned in Lillehammer (in connection with the winter Olympics in 1994), that a combined pre- and post-denitrification process should be used. This process would take advantage of the benefits of both the pre-denitrification system (utilization of easily biodegradable organic matter in the raw water) and the post-denitrification system (securing the treatment result without being dependent on the raw water characteristics). Even at very low temperatures during the start-up period of Lillehammer WWTP, the results were remarkably good, resulting in 82 % removal of inorganic N at 6-6.5 °C. Of the 82 %, only 15-17 % was removed in the pre-denitrification stage because of the cold, snow-melt diluted wastewater.

5.3 The full-scale plants survey

The Lillehammer experiences led to the use of the combined pre- and post-denitrification process being used at also other full-scale plants in Norway, like Nordre Follo-, Nedre Romerike- and Gardermoen wastewater treatment plants that constitutes the participants of the operational survey in chapter 4 of the report.

Lillehammer WWTP. Data for 2018 and 2019 were presented. The plant was at this time much higher loaded than designed for and the plant owner (Lillehammer city) now used the plant as a pure post-denitrification plant.

In 2018 and 2019 the plant had an average removal of 79.7 % and 75.1 % of total N respectively, well above the required 70 %, at an average temperature of around 10 °C and a minimum temperature of 3.9 and 5.3 °C respectively. The nitrification rate as calculated over the nitrification reactors (R4 to R6) was in the range pf 1.0 – 1.2 g NH_4-N/m^2d in the temperature range of 5°C – 10°C (see Figure 5.1) both in 2018 and 2019 with a relatively small variation within this temperature range.

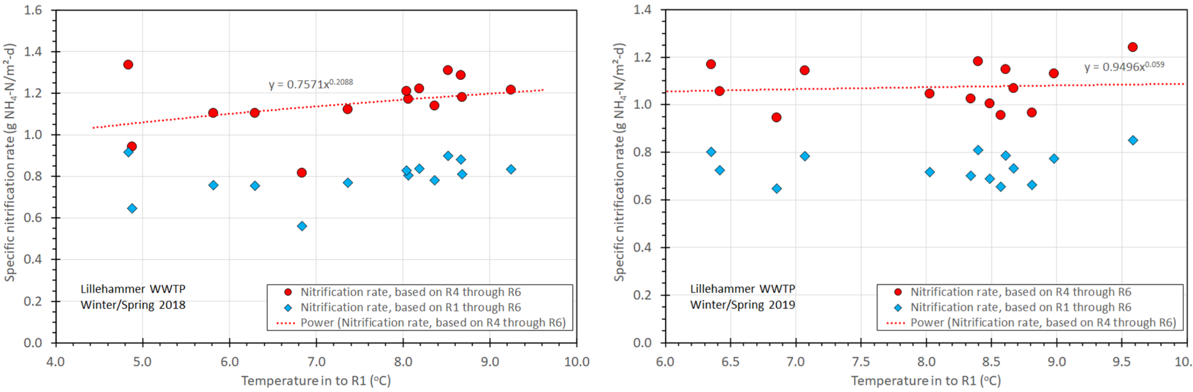


Figure 5.1. Specific nitrification rates versus temperature, based on weekly averages in the winter of 2018 (left) and 2019 (right).

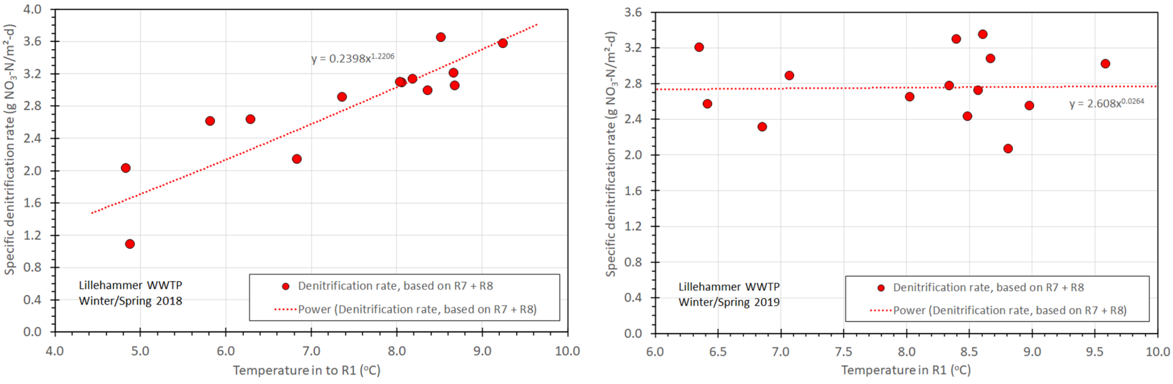


Figure 5.2. Specific post-denitrification rates versus temperature, based on weekly averages from in winter and spring 2018 (left) and 2019 (right).

Figure 5.2, that show the post-denitrification rates, appears to indicate a very high temperature effect in 2018. However, at temperatures below 8 °C the effluent concentrations were steady around 4-6 mg NO_3-N/l, and the significantly reduced rates at the lowest temperatures were mainly due to the large reduction in influent concentrations caused by the large increase in the influent flow rates. In 2019 there was no visible temperature effect on the post-denitrification rate (around 2.8 g NO_x-N/m^2d) in the temperature interval from 6.3 to 9.6 °C. The high rate is caused by the fact that ethanol is used as external carbon source in this plant. Comparing the results from 2019 with the results from 2018, it appears that the much higher flows at low temperatures in 2018 had a higher impact on the post-denitrification rates than the temperature itself.

Nordre Follo WWTP. Data for 2019 and 2020 were presented and discussed. The plant had an average removal of 79.8 % and 78.5% of total N respectively at an average temperature of 11.3 and 11.8 °C respectively and a minimum temperature of around 7.5°C. This was well above the required 70 % total N-removal, in spite of an 80-percentile total N load that was 26 % and 33 % higher respectively than the design load.

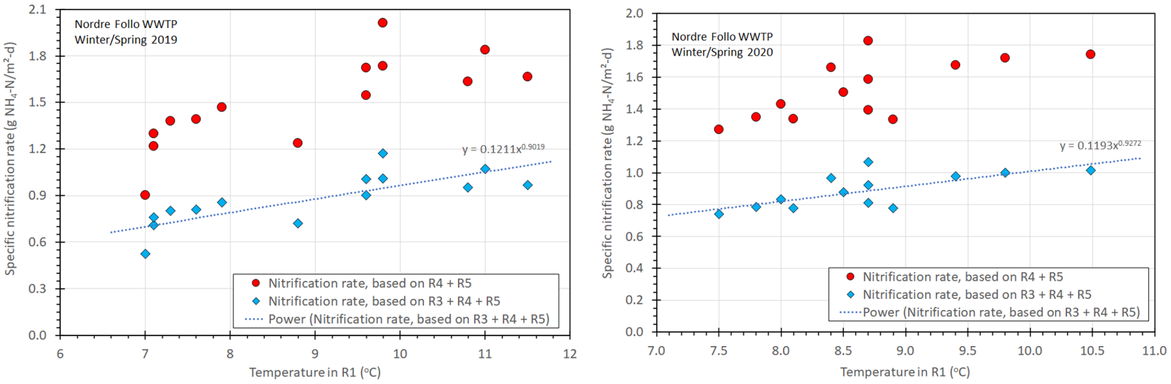


Figure 5.3. Specific nitrification rates versus temperature, based on weekly averages in winter and spring of 2019 (left) and 2020 (right).

A mass balance analysis gave an average pre-denitrification rate of 0.60 g NO_3-N/m^2-d (2019) and 0.70 g NO_3-N/m^2-d (2020). This fairly high pre-denitrification rate is explained by a possible boost by methanol added to the MBBR influent.

Nedre Romerike Avløpsselskap (NRA) WWTP. The data analyzed for NRA are those with the lowest temperatures over a period of 2-4 weeks for the 4 years 2013, 2014, 2016 and 2018.

The nitrification rates and denitrification rates over these years are summarized in Table 5.1. This plant has not a particular BOD-removal reactor, so the nitrification rates are calculated based on all the aerobic reactors (including de-ox reactor) that remove BOD as well as ammonium. Methanol is normally used as external carbon source.

Table 5.2. Average temperatures, nitrification rates and denitrification rates over weeks of low temperatures in the years 2013, 2014, 2016 and 2018 at NRA WWTP based on on-line measurements in the plant.

	2013	2014	2016	2018	Average
Average temperature,°C	8.9	8.8	8.7	7.9	8.6
Minimum temperature, °C	6.6	6.4	7.4	5.2	5.2
Maximum temperature, °C	10.4	11.2	9.7	10.2	11.2
Aver. nitrification rate (g NH_4-N/m^{2}d)	0.31	0.41	0.39	0.36	0.37
Aver. pre-denitrification rate (g NO_x-N/m^{2}d)	0.18	0.20	0.18	0.22	0.20
Aver. post-denitrification rate (g NO_x-N/m^{2}d)	1.11	1.07	1.03	1.00	1.05

Roughly these data show an Arrhenius temperature correction value of 1.07 – 1.11 both for nitrification and denitrification in the temperature range of 5 – 10 °C. These are calculated over relatively short periods and should be taken with caution.

Gardermoen WWTP. Data from 2019 and 2020 were analyzed. Treatment efficiencies were significantly better than required in both years, with average removal of about 85 % total N. Overall, the results were excellent for a plant that in 2019 received a load of total N and organic matter that was 1.5 to 1.8 times higher than the design load. In 2020 loads were lower caused by the pandemic that reduced the activity at the Oslo airport for the last 8-9 months of the year. In a normal year, more than 20 % of the total N to the treatment plant originates from the airport.

Effluent NH_4-N concentrations were most of the time very low, and hence limiting with respect to nitrification rates. Nitrification rates over the reported period are shown in Figure 5.4 and nitrification rates versus temperature in Figure 5.5.

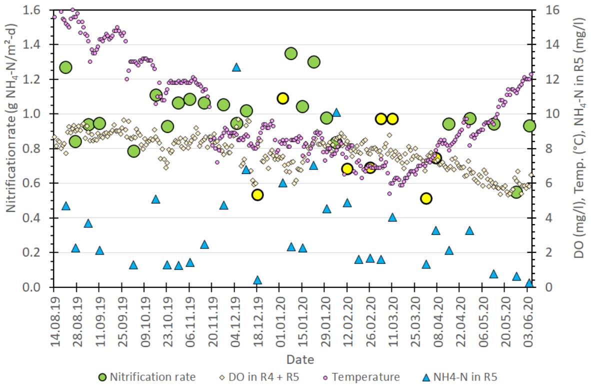


Figure 5.4. Nitrification rates in MBBRs in train 2 at GRA. Rates shown with yellow circles are from a period with only R1 anoxic in train 2. Many of the rates were limited by low NH_4-N concentrations.

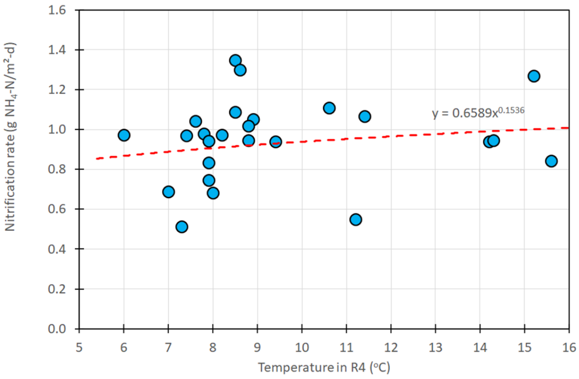


Figure 5.5 Nitrification rates in MBBRs in train 2 at GRA versus temperature in R4. Rates shown only for days when nitrification was not limited by low NH_4-N concentrations.

There is a large spread in Figure 5.5 since so many factors influence on nitrification rate (DO being the most important one). However, the trend line indicates a very low temperature dependency with an apparent θ of only 1.015 over the temperature range from 6 to 16 °C. As mentioned previously, increased DO at low temperatures can mask the true temperature dependency.

Figure 5.6 shows the pre-denitrification rates and Figure 5.7 the post-denitrification rates together with C/N-ratios and temperatures at GRA during the period of analysis.

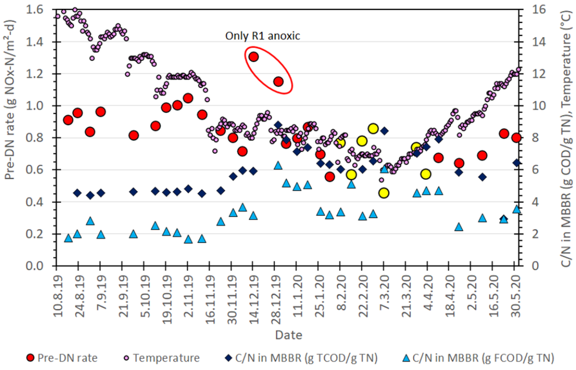


Figure 5.6. Pre-denitrification rates in MBBRs at GRA.

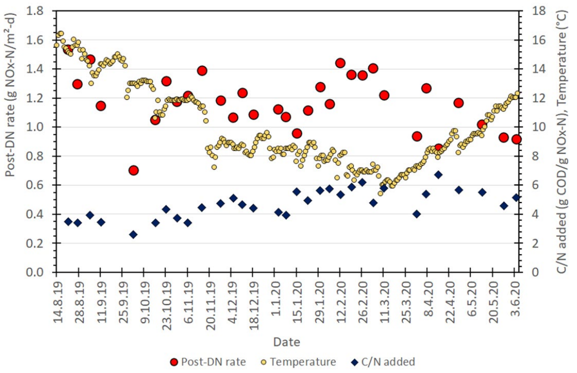


Figure 5.7 Post-denitrification rates in R6, train 2, at GRA.

Many factors influence the pre-denitrification rates, so it is difficult to single out the temperature effect. However, the data indicate reduced rates when the temperature dropped below 8 °C. In the temperature range from 8.0 to 16.5 °C the average pre-denitrification rate was 0.88 g NOx-N/m²d. This dropped to an average of 0.68 g NOx-N/m²d in the temperature range from 5.4 to 8.0 °C.

No temperature dependency can be found in the post-denitrification reactor. Some of the highest post-denitrification rates were in fact seen at the very lowest temperatures. Most probably this was due to the operators increasing the C/N ratio to make up for the slightly reduced pre-denitrification rates at the lowest temperatures. At 7 °C the post-denitrification rates were about 1.4 g NOx-N/m²-d, which is the maximum you can expect with MPG as external carbon source. The average post-denitrification rate for all the data points in Figure 5.7 was 1.2 g NOx-N/m²-d.

5.4 Design of MBBR for N-removal

Norsk Vann has published design guidelines for wastewater treatment plants since 1978 (revised in 1983, 2009 and 2020). The MBBR was included in the 2009-revision based mainly on the experiences presented in chapter 3 in this report. Later more full-scale experiences from plants like Lillehammer-, Nordre Follo- , Nedre Romerike-, and Gardermoen wastewater treatment plants gave experiences that constituted the basis for the more comprehensive design guidelines in the present revision (Norsk Vann, 2020). All of these plants experience periods with cold wastewater well below 10°C – sometimes below 5°C.

Based on the compilation of experiences from research papers (chapter 3) as well as collected full-scale data from recent years (chapter 4), it can be concluded that these guidelines (on design of MBBR) are well suited for designing wastewater treatment plants that experience cold water situations in Northern Sweden - of the same kind as experienced in the Norwegian plants.

The effluent standard in Sweden will most probably be set at 10 mg Tot N/l (as yearly average) or 70 % removal as yearly average – as it is in Norway. In the spring situation, it will be much easier to reach the concentration standard (because of the diluted water in) than the % removal standard, We recommend, for the cold water plants in Sweden, to use the Norwegian design quidelines for MBBR and design for a temperature of 10 ^{o}C, and check the volumes for the situations of the minimum temperature (and corresponding high flow). An excerpt of the Norwegian design guidelines that contains the content regarding design of MBBR-plants is included in this report as an attachment. Please refer to the full Norwegian design guidelines (Norsk Vann, 2020) for info on design loadings etc.

We recommend to use the combined pre- and post-denitrification process with great flexibility (see flow-sheet below), i.e. the use of swing reactors and advanced steering equipment regulated through DO, ammonium and/or nitrate sensors. This would be DO-control to regulate air supply in nitrifying reactors and hence nitrification rate and external carbon addition control to regulate denitrification rate.

In the figure below we have indicated the possible use of pre-coagulation in order to increase capacity of particulate BOD-removal before the MBBR, and hence nitrification capacity in the MBBR. We have also proposed to use flotation for final particle separation. By this a re-ox stage may be omitted.

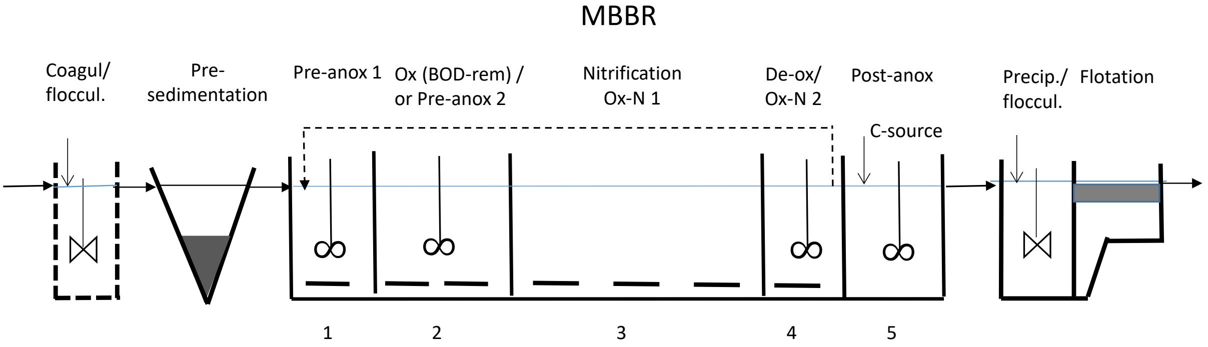


Figure 5.8 Recommended flow diagram for cold wastewater nutrient removal plants in Sweden

6 References (listed after year of publication)

Ødegaard, H. (1992) Fjerning av næringsstoffer ved rensing av avløpsvann. TAPIR Forlag, 1992. ISBN 82-519-1109-5 (in Norwegian)

Hem, L., Rusten, B,, Ødegaard, H. (1994) Nitrification in a Moving Bed Biofilm Reactor. *Water Research*, Vol. 28, No. 6, pp 1425-1433

Rusten B., Siljudalen J. G. and Nordeidet B. (1994) Upgrading to nitrogen removal with the KMT moving bed biofilm process. *Water Science and Technology*, Vol. 29, No. 12, pp. 185-195.

Rusten, B., Siljudalen, J. G. and Bungum, S. (1995a). Moving bed biofilm reactors for nitrogen removal – From initial pilot testing to start-up of the Lillehammer WWTP. Proceedings, WEFTEC'95, Miami Beach, Florida, USA, Oct. 21-25, Volume 1, pp. 615–626.

Rusten, B., Hem, L.J. and Ødegaard, H. (1995b). Nitrification of municipal wastewater in moving-bed biofilm reactors. *Water Environ. Res*., 67, 75–86.

Rusten, B., Hem, L.J. and Ødegaard, H. (1995c). Nitrogen removal from dilute wastewater in cold climate using moving-bed biofilm reactors. *Water Environ. Res*., 67, 65–74.

Rusten, B., Wien, A. and Skjefstad, J. (1996). Spent aircraft deicing fluid as external carbon source for denitrification of municipal wastewater: From waste problem to beneficial use. 51st Purdue Industrial Waste Conference Proceeding, Ann Arbor Press, Inc., Chelsea, Michigan, USA, pp.359–369.

Andreottola, G.; Foladori ,P.; Ragazzi, M. (2000) Upgrading of a small wastewater treatment plant in a cold climate region using a moving bed biofilm reactor (MBBR) system. *Water Sci Technol*. 41 (1): 177–185. https://doi.org/10.2166/wst.2000.0027

Rusten, B.; Hellström, B.G., Hellström, F.; Sehested, O.; Skjelfoss, E.; Svendsen, B. (2000) Pilot testing and preliminary design of moving bed biofilm reactors for nitrogen removal at the FREVAR wastewater treatment plant. *Water Science and Technology*, Vol 41 No 4–5 pp 13–20

Welander, Mattiasson (2003) Denitrification at low temperatures using a suspended carrier biofilm process. *Water Research*, 37 (2003) 2394–2398

Salvetti, R.; Azzellino, A.; Canziani, R.; Bonomo, L. (2006) Effects of temperature on tertiary nitrification in moving-bed biofilm reactors. *Water Research,* 40, 2981 – 2993

Wessman, F.G. and Johnson, C. H. (2006) Cold weather nitrification of lagoon effluent using a moving bed biofilm reactor (MBBR) treatment process. Proceedings of the Water Environment Federation, 2006 /14, WEFTEC 2006

Rusten B. and Ødegaard H. (2007) Design and Operation of Moving Bed Biofilm Reactor Plants for Very Low Effluent Nitrogen and Phosphorus Concentrations. *Water Practice*, Vol. 1, No. 5, November 2007.

Delatolla R, Tufenkji N, Comeau Y, Gadbois A, Lamarre D,Berk D. (2009) Kinetic analysis of attached growth nitrification in cold climates. (2009) *Water Sci Technol*., 60, 1173–1184.

Delatolla R, Tufenkji N, Comeau Y, Gadbois A, Lamarre D, Berk D. (2010) Investigation of laboratory-scale and pilot-scale attached growth ammonia removal kinetics at cold temperature and low influent carbon. *Water Qual Res J Can*., 45, 427–436.

Eriksson, J. (2011) The influence of the nitrate load on the denitrification capacity - performed at GRYAABs MBBR plant. Eksamensarbete kemiingeniörsprogrammet, Chalmers tekniska högskola, Gothenburg 2011.

Hoang V, Delatolla R, Abujamel T, Mottawea W, Gadbois, A, Laflamme E, Stintzi, A. (2014) Nitrifying moving bed biofilm reactor (MBBR) biofilm and biomass response to long term exposure to 1◦C. *Water Research* 49, 215–224.

Almomani, F.A., Delatolla, R. and Örmeci, B. (2014) Field study of moving bed biofilm reactor technology for post-treatment of wastewater lagoon effluent at 1°C, *Environmental Technology*, 35:13, 1596-1604, DOI: 10.1080/09593330.2013.874500

Young, B., Delatolla, R., Ren, B., Kennedy, K., Laflamme, E., Stintzi, A. (2016) Pilot-scale tertiary MBBR nitrification at 1°C: characterization of ammonia removal rate, solids settleability and biofilm characteristics, *Environmental Technology*, 37:16, 2124-2132, DOI: 10.1080/09593330.2016.1143037

Young, B. (2017) Nitrifying MBBR performance optimization in temperate climates through understanding biofilm morphology and microbiome. PhD thesis, Ottawa-Carleton Institute for Environmental Engineering, Department of Civil Engineering, University of Ottawa, Canada

Ahmed, W., Tian, X. & Delatolla, R. (2019 Nitrifying moving bed biofilm reactor: Performance at low temperatures and response to cold-shock. *Chemosphere*. 229, 295-3102. https://doi.org/10.1016/j.chemosphere.2019.04.176

Figdore, B. et al (2019) Adaptive facility plan and upgrades provide capital savings resiliency and cold weather ammonia removal in Palmer, Alaska. Proc. 92nd Water Environment Federation Technical Exhibition and Conference (WEFTEC 2019), Chicago, Illinois, USA, 21 - 25 September 2019.

Norsk Vann (2020) Veiledning for dimensjonering av avløpsrenseanlegg. Rapportnummer: 256/2020 ISN 978-82-414-0450-4

7 Attachment – (Excerpt of the Norwegian design guidelines on wastewater treatment plants (Norsk Vann, 2020 – In Norwegian)

MBBR

Moving Bed Biofilm Reaktorer (MBBR) er en biofilmreaktor med gjennomstrømning og ingen slamretur. Anlegg basert på MBBR er sammensatt av MBBR reaktor og slamseparasjonsrektor (evt. med felling og flokkulering).

Beskrivelse

I MBBR-baserte prosesser vokser biomassen på plastelementer som holdes svevende i reaktoren ved hjelp av lufting eller omrøring.

Plastelementene (biofilmbærerne) hindres i å forlate reaktoren sammen med vannet ved en silanordning på utløpet av reaktoren. Når MBBR brukes for aerob, biologisk nedbrytning av organisk stoff eller ammonium, holdes bærerne suspendert og totalomrørt i reaktoren gjennom lufting som er nødvendig for den biologiske omsetning. Når MBBR brukes for anoksiske prosesser (for biologisk nedbrytning av nitritt og nitrat) eller for anaerobe prosesser, holdes bærerne suspendert og totalomrørt i reaktoren gjennom mekanisk omrøring.

Ved behandling av kommunalt avløpsvann benyttes vanligvis forbehandling gjennom rist og sandfang samt primærrensing gjennom finsil, forsedimentering eller forfelling. Dersom det tas tilstrekkelig hensyn til det (finrist, tilstrekkelig lysåpning på reaktorsiler etc.) kan primærsteget utelates om ønskelig.

Anvendelsesområde

MBBR-baserte prosesser kan benyttes til alle formål der det er behov for biologisk behandling av avløpsvann, dvs. fjerning av organisk stoff, nitrifikasjon, nitrogenfjerning og biologisk fosforfjerning. Foreløpig er imidlertid biologisk fosforfjerning i MBBR-anlegg på utviklingsstadiet og derfor gis det i denne veiledningen ikke dimensjoneringsdata for denne anvendelsen.

I de fleste MBBR anlegg kombineres biologisk og kjemisk rensing ved at felling/koagulering og flokkulering settes inn direkte etter bioreaktor og før slamseparasjonsreaktor (kalt Biofilm m/felling i tabell 1.4.1). Dersom etterfelling benyttes i MBBR-anlegg (dvs anlegg med eget separasjonssteg for biofilmslammet), er separasjonsmetoden i etterfellingssteget normalt mikrosiling eller filtrering (etter felling/koagulering og flokkulering).

Dersom man benytter MBBR i kombinasjon med kjemisk felling, vil man få bedre separasjon av den finpartikulære slamfraksjonen. Derfor kan man tillate en høyere organisk belastning for å oppnå en ønsket renseeffekt i anlegg basert på biofilm m/felling.

Høyt belastede MBBR-anlegg kan også benyttes som første trinn i en to-trinns biologisk prosess. Dette kan f.eks. være aktuelt ved høyt innhold av løst organisk stoff, f.eks. som et resultat av høy industribelastning. Det kan også være hensiktsmessig i et for-renseanlegg der man bare ønsker fjerning av lett nedbrytbart organisk stoff (dvs. liten eller ingen hydrolyse av partikulært organisk stoff), som f.eks. forbehandling i anlegg for de-ammonifikasjon. I slike høyt belastede anlegg blir ofte den hydrauliske belastningen på silene dimensjonerende.

For anlegg med høyt innhold av løst organisk stoff, f.eks. som et resultat av høy industribelastning, er også biologisk fosforfjerning aktuelt.

Dimensjonering av bioreaktor

Dersom man benytter forsedimentering eller forfelling som forbehandling i MBBR-anlegg, skal stoffbelastningen inn på det biologiske steget reduseres tilsvarende det som er angitt i pkt 3.5.1.

Dersom man benytter anlegg basert på biofilm m/felling skal bioreaktoren dimensjoneres på samme måte som angitt under for biologisk rensing med MBBR.

Nødvendig reaktorvolum

Reaktorvolumet bestemmes på grunnlag av dimensjonerende arealbelastning i g/m$^2 \cdot$ d, fyllingsgraden av bæremedium (%) og effektivt, spesifikt biofilmareal på bæremediet (m^2/m^3). I tabell 3.5.6 er gitt verdier for dimensjonerende arealbelastning.

Tabell 3.5.6. Dimensjonerende verdier for MBBR-reaktorer ved 10 °C.

Behandlingsmålsetting	**Organisk arealbelastning g BOF$_5$/m$^2 \cdot$ d**	**Ammonium arealbelastning g NH$_4$-N/m$^2 \cdot$ d**	**NO$_3$-N$_{ekvivalenter}$ arealbelastning g NO$_3$-N/m$^2 \cdot$ d**
A. Hoveddelen av BOF$_5$ skal fjernes: > 70 % fjerning eller < 25 mg BOF$_5$/l i utløp			
• Uten kjemikalie tilsetting	≤ 5[1]		
• Med polymerkoagulering	≤ 8[1]		
• Med kjemisk etterfelling	≤ 11,5[1]		
B. I tillegg til A: Hoveddelen av NH$_4$-N skal nitrifiseres: • < 3 mg NH$_4$-N/l (får dermed < 15 mg BOF$_5$/l)	≤ 5[2]		
a. Uten forsed.el. for-denitrfik.		≤ 0,50[3]	
b. Med forsed,el. for-denitrfik.		≤ 0,60[3]	
c. Med forsed,og for-denitrfik.		≤ 0,65[3]	
d. Med forfelling		≤ 0,75[3]	
C. I tillegg til A og B: Hoveddelen av tot N skal fjernes (>70 %, får dermed <10 mg BOF$_5$ i utløp):			
• Med for-denitrifikasjon			≤ 0,50[4,5]
• Med etter-denitrifikasjon			≤ 1,50[6]
• Med kombinert for- og etterdenitrifikasjon			≤ 0,50[4,5] i for-DN-reakt. ≤ 1,50[6] i etter-DN-reakt.

[1] *Det forutsettes at reaktoren er inndelt i minst to adskilte kammer (angitt verdi gjelder totalvolum), og at oppholdstiden ved* $Q_{maksdim}$ *er* $\geq$*30 min.*

[2] *Dimensjoneringsverdi for den reaktordelen som forutsettes å fjerne organisk stoff før nitrifikasjonen*

[3] *Dimensjoneringsverdi for den reaktordelen som forutsettes å fjerne ammonium ved nitrifikasjon forutsatt at restkonsentrasjonen av ammonium er* $\geq$ *2 mg* NH_4*-N/l. Ved behov for lavere restkonsentrasjon skal de angitte nitrifikasjonshastigheter reduseres lineært fra angitt verdi ved* NH_4*-N konsentrasjon lik 2 mg* NH_4*-N/l til 0 ved* NH_4*-N konsentrasjon lik 0 mg* NH_4*-N/l. Det skal ikke forutsettes høyere oksygenkonsentrasjon enn 5 mg* O_2*/l ved dimensjonering. De angitte nitrifikasjonshastighetene gjelder ved oksygenkonsentrasjon = 5 mg* O_2*/l*

[4] *Forutsatt at C/N-forhold i innkommende vann til fordenitrifikasjons-reaktoren er* $\geq$ *4 g* BOF_5*/g* NO_3*-*$N_{ekviv,\ inn}$ *og restkonsentrasjonen kan være* $\geq$ *3 mg* NO_3*-N/l. Dersom C/N-forholdet er lavere enn dette (som er normalt), skal hastigheten reduseres lineært fra angitt verdi ved C/N = 4* $BOF_{5,inn}$*/g* NO_3*-*$N_{ekviv.fjernet}$*. til 0 ved C/N = 2 g* $BOF_{5,inn}$*/g* NO_3*-*$N_{ekviv.fjernet}$*. Er det to fordenitrifikasjons reaktorer i serie, skal den første dimensjoneres som anvist og den siste for* $\leq$ *0,30 g* NO_x*-N/*m^2*d*

[5] *Forutsatt at resirkulert vannmengde fra nitrifikasjons- til fordenitrifikasjonsreaktor er* $\geq$*250 % av* $Q_{midl,\ inn}$ *dersom man skal klare kravet om 70 % fjerning med fordenitrifikasjon alene. Ved kombinert for- og etterdenitrifikasjon kan resirkuleringsmengden være lavere.*

[6] *Forutsatt tilsatt ekstern karbonkilde tilsvarende C/N-forhold i innkommende vann til etterdenitrifikasjonsreaktoren* $\geq$ *3 g* BOF_5*/g* NO_3*-*$N_{ekviv.}$ *(evt. 4,5 g KOF/g* NO_3*-*N_{ekviv}*) og restkonsentrasjonen av* NO_3*-*$N_{ekviv.}$ *kan være* $\geq$ *3 mg* NO_3*-N/l. Er det behov for en lavere restkonsentrasjon, skal arealbelastningen for etterdenitrifikasjonen multipliseres med en faktor F. F= nødvendig konsentrasjon/3 . Disse verdiene gjelder når metanol eller glykol benyttes som ekstern karbonkilde. Dersom etanol benyttes, kan hastighetene multipliseres med 1,8.*

Ved andre temperaturer enn 10 °C skal arealbelastningen korrigeres etter følgende formel:

$$A_T = A_{10} \cdot \Theta^{(T-10)} \qquad \text{lign. 3.19}$$

hvor

- A_T er arealbelastning ved den aktuelle temperatur
- A_{10} er arealbelastning ved 10 °C.
- Θ = Temperaturkorreksjonsfaktoren , som skal settes til:
  - BOF_5-fjerning og denitrifikasjon: 1,07
  - Nitrifikasjon: 1,09

Høybelastet MBBR for fjerning av løst organisk stoff i for-renseanlegg (60-70 % BOF_5-fjerning og 25-40 mg BOF_5 i effluenten) kan dimensjoneres for 15 g $BOF_{5,løst}/m^2d$ (ved 10 °C). Ved høyere temperaturer skal man aldri dimensjonere for høyere belastning enn 25 g $BOF_{5,løst}/m^2d$. Ofte blir reaktor-sil kapasiteten dimensjonerende for akseptabel vannmengde i slike anlegg.

Anlegg for nitrogenfjerning kan bygges med fordenitrifikasjon, etterdenitrifikasjon eller en kombinasjon av de to. Den delen av anlegget som skal sørge for fjerning av organisk stoff og nitrifikasjon, dimensjoneres i henhold til behandlingsmålsetting B, mens denitrifikasjonsbassengene dimensjoneres etter behandlingsmålsetting C.

I fordenitrifikasjonsanlegg skal man bestemme nødvendig volum for fjerning av organisk stoff, utover opptaket (fjerningen) i fordenitrifikasjonen, basert på den gjenværende $BOF_{5,tot}$ etter fordenitrifikasjonssteget

For bestemmelse av omsetning av organisk stoff ved fordenitrifikasjon, skal man forutsette (dersom man ikke har data som viser noe annet):

a. Andel løst organisk stoff i innløpsvann i forhold til total ($BOF_{5,løst}/BOF_{5,total}$) :
   - Uten primærrensetrinn: 25%
   - Med primærrensetrinn: 30 %

b. Andel partikulært organisk stoff som blir omsatt til løst organisk stoff ved hydrolyse: 25 % av $BOF_{5,partikulær}$ i innløpsvann

c. Omsetning av organisk stoff ved fordenitrifikasjon: 3,0 g $BOF_{5,løst}/gNO_{3\ ekviv.}\text{-}N_{fjernet}$ hvor løst organisk stoff tilgjengelig for denitrifikasjon ($BOF_{5,løst}$) utgjøres av summen av de to fraksjoner under punkt a og b over

Belastningen (kg NO_3-N/d) på fordenitrifikasjonsreaktoren skal bestemmes som produktet av den returnerte vannmengden og konsentrasjonen av nitrat i utløpet av nitrifikasjonssonen - med tillegg av den konverterte oksygenmengden (1 g oksygen tilsvarer 0,35 g $NO_3\text{-}N_{ekvivalenter}$).

I anlegg hvor fordenitrifikasjon inngår vil det derfor være hensiktsmessig å redusere oksygeninnholdet i en deoksygeneringsreaktor før det nitratholdige vannet returneres til fordenitrifikasjonsreaktoren. Dette kan gjøres ved å la være å lufte denne reaktoren (benytte omrører) slik at biomassen på bærerne forbruker oksygen som følge av nedbrytningen av gjenværende ammonium. Deoksygeneringsreaktoren kan dimensjoneres basert på en nedbrytningshastighet på 0,225 g NH_4-N/m^2d (ved 10^0C – temperaturkoeffisient $\Theta = 1{,}09$ – se lign. 3.19). Man kan da anta at oksygenkonsentrasjonen i vannet som skal resirkuleres blir 2 g O_2/m^3.

Tilsvarende vil det, i anlegg hvor etterdenitrifikasjon inngår, være hensiktsmessig med en deoksygeneringsreaktor for å redusere oksygenbelastningen på etterdenitrifikasjonen. Det vil også være hensiktsmessig å benytte en luftet reoksygeneringsreaktor etter etterdenitrifikasjonsreaktoren for å sikre at eventuelt gjenværende organisk stoff fra den eksterne karbonkilde blir nedbrutt. Denne reoksygeneringsreaktoren kan dimensjoneres for en restkonsentrasjon av organisk stoff på 10 g $KOF_{løst}/m^3$ og en nedbrytningshastighet på 4 g $KOF_{løst}/m^2\cdot d$ ved $Q_{maksdim}$. Oppholdstiden skal dog ikke settes lavene enn 18 min ved $Q_{maksdim}$.

Nødvendig lufttilførsel

Oksygenbehovet (OB, kg O_2/time) skal dimensjoneres på grunnlag av:

a. Behandlingsmålsetting A: OB = 1,0 kg O_2/kg $BOF_{5,\ tilført}$.
    - Ved bestemmelse av oksygenbehov (kg O_2/time) skal det legges inn en spissbelastningsfaktor på 1,3.

b. Behandlingsmålsetting B: OB = 1,0 kg O_2/kg $BOF_{5,\ tilført}$ + 4.3 kg O_2/kg $NH_4\text{-}N_{tilført}$
    - For behandlingsmålsetting B legges inn en spissbelastningsfaktor på 2,0 for oksygenbehov (kg O_2/time) til nitrifisering. Det legges da ikke inn spissbelastningsfaktor for oksygen til fjerning av organisk stoff, ettersom spissbelastning av organisk stoff og nitrogen ikke vil opptre samtidig.

Den dimensjonerende oksygentilførsel (OT) er avhengig av oksygenbehovet (OB), innblåsingsdybden, forbehandlingen etc. og bestemmes på samme måte som i aktivslamanlegg (lign. 3.5.15).

I MBBR-anlegg er den spesifikke oksygenoverføringskapasiteten (SOFK) i tillegg til å være avhengig av luftesystemets evne til å overføre oksygen fra boble til luft, samt innblåsingsdypet, også avhengig av utforming og fyllingsgrad av mediet. Dersom ikke noe annet tilsier det, kan verdiene i tabell 3.5.6 være veiledende ved overslagsberegninger:

Tabell 3.5.7. Veiledende spesifikk oksygenoverføringskapasitet (SOFK) i MBBR-anlegg med diffusorsystem basert på borede 4mm hull med middels boblestørrelse.

Type av biofilm bærer	**Fyllingsgrad (%)**	**SOFK$_{avløpsvann}$ (g O_2/Nm$^3_{luft}$*m) (10^{o}C, 1 atm)**
Rør-lignende	30 (~20-40) 50 (~40-60)	6,5-7,5 7,5-8,5
Mynt-lignende	30 (~20-40) 50 (~40-60)	5,5-6,0 6,0-6,5

Luftmengden beregnes som følger:

$$Q_{luft}\ (m^3/time) = \frac{OT\ (kgO_2/time)\cdot 1000\ (g/kg)}{SOFK\ (gO_2/Nm^3_{luft}\cdot m_{innblåsningsdyp})\cdot h_d\ (m_{innblåsningsdyp})} \qquad \text{lign. 3.5.20}$$

I enkelte tilfeller kan nødvendig grad av omrøring bli dimensjonerende for lufttilførselen. For å sikre god omrøring bør lufttilførselen aldri settes lavere enn 10 - 15 m$^3_{luft}$/h · m$^2_{bunnflate\ i\ reaktor}$ (avhengig av bærer).

Slamproduksjon

For bestemmelse av slamproduksjon skal følgende verdier benyttes (ved 10^{o}C)

- Ved fjerning av BOF$_5$ og for-denitrifikasjon:
    - Uten forsedimentering: 1,15 g TS/g BOF$_{5,fjernet}$
    - Med forsedimentering: 1,00 g TS/g BOF$_{5,fjernet}$
    - Med forfelling: 0,85 g TS/g BOF$_{5,fjernet}$
- Tillegg ved nitrifikasjon: 0,125 g TS/g NH_4-N fjernet
- Tillegg ved etter-denitrifikasjon med ekstern karbonkilde: 0,60 g TS/g BOF$_{5,fjernet}$

Biologisk fosforfjerning

Biologisk fosforfjerning krever at råvannet har et høyt innhold av lett tilgjengelig organisk stoff i forhold til fosforinnholdet. Under pkt. 3.5.2. Aktivslamprosessen (Biologisk fosforfjerning) er dette nærmere beskrevet.

Ved biologisk fosforfjerning må biomassen gjennomgå vekselsvis anaerobe og aerobe forhold og derfor er anlegg for biologisk fosforfjerning vanligvis basert på aktivslamprosessen. Skal en biofilmprosess benyttes (f.eks. MBBR), må man enten benytte satsvis drift, eller flytte biomediet (med biomassen på) fra reaktorseksjoner uten lufting (anaerobe forhold) til reaktorseksjoner med lufting (aerobe forhold). Slike metoder er under utvikling, men det finnes ikke tilstrekkelig med full-skala erfaringer, basert på et bredt utvalg avløpsvann med forskjellig karakteristikk, som kan danne grunnlag for å angi dimensjoneringsdata for disse prosessløsningene, på nåværende tidspunkt.

For dimensjonering av patenterte MBBR-systemer for biologisk fosforfjerning henvises det til de aktuelle leverandørene.

Dimensjonering av slamseparasjonsreaktorer

Generelt

Når det forekommer krav om fjerning av fosfor i tillegg til organisk stoff (og evt. nitrogen) skjer dette vanligvis ved kjemisk felling og flokkulering direkte etter bioreaktoren og før slamseparasjonen - også kalt biofilm m/felling (se tabell 1.1).

Forskjellen til tradisjonell etterfelling er at det ved etterfelling er et eget, separat slamseparasjonsbasseng for bioslam etter bioreaktoren og et nytt etter fellingssteget, mens det i anlegg med MBBR m/felling kun er ett for bioslam og kjemisk slam samlet. Dette er mulig fordi slamkonsentrasjonen ut av MBBR-reaktoren typisk er bare ca 200 mg SS/l mens den ut av en aktivslamreaktor typisk er ca 3-4000 mg SS/l (ved behandling av kommunalt avløpsvann).

Alle typer av slamseparasjonsreaktorer som er omtalt i denne veiledningen, kan benyttes til separasjon av slam fra MBBR-reaktorer – med eller uten felling.

Separasjon i finsil- eller mikrosilanlegg direkte etter MBBR, kan vise seg mindre hensiktsmessig dersom man vil måtte benytte høy polymerdosering (evt. i tillegg til metall-dosering for å felle ut fosfor). Denne løsningen kan likevel ikke helt utelukkes. Derimot kan mikrosilanlegg benyttes for polering etter annet hovedseparasjonssteg (se avsnitt 3.6).

Normalt benyttes flokkulering foran slamseparasjonssteget. Flokkuleringen dimensjoneres på samme måte som i kjemiske anlegg (se pkt. 3.4.3).

Normalt benyttes sedimentering, lamellsedimentering eller flotasjon for slamseparasjon i MBBR-anlegg (med eller uten felling). Dersom etterfelling basert på MBBR benyttes, skal flokkuleringsbassenget dimensjoneres som angitt i pkt. 3.4.3.

Slamseparasjon ved sedimentering

Konvensjonelle sedimenteringsbasseng for MBBR-anlegg (og øvrige biofilmanlegg) skal dimensjoneres mht. overflatebelastning i henhold til tabell 3.5.8.

Tabell 3.5.8. Dimensjoneringsdata for konvensjonelle sedimenteringsbasseng etter MBBR.

Sedimenterings-bassengenes funksjon		**Vanndyp[1)] m**	**Overflatebelastning m³/m² · time**	
			Q_{dim}	**$Q_{maksdim}$**
Sluttsedimentering uten kjemisk felling	Etter MBBR (og andre biofilmreaktorer)	2,5 ≥ 3,0	0,8 1,0	1,1 1,6
Sluttsedimentering ved MBBR m/felling	Etter MBBR (og andre biofilmreaktorer) som etterfølges av felling/flokkulering	2,5 ≥ 3,0	1,0 1,3	1,6 2,0
Sluttsedimentering ved etterfelling	Etter MBBR m/egen separasjon	2,5 ≥ 3,0	1,0 1,3	1,6 2,0

[1)] For spissbunnede basseng uten slamskrape skal flaten på 1 meters vanndyp oppfylle kravene. Som vanndyp regnes den totale vanndybde fratrukket 1,0 m, som slamsone.

Dersom man bruker polymer som flokkuleringsmiddel, kan man oppnå høyere sedimenteringshastigheter på fnokker enn uten bruk av polymer. Verdiene i tabell 3.5.7 er gitt for en situasjon der polymer ikke benyttes. Det er vanskelig å anslå hvor stor forbedringen i sedimenteringshastighet vil kunne bli som følge av polymertilsettingen, og det gis derfor ikke spesifikke anvisninger for dette. Det anbefales at det gjøres forsøk i hvert enkelt tilfelle. For overslagsberegninger kan man anta at overflatebelastningen kan økes med 0,5 $m^3/m^2 \cdot$time i forhold til de verdier som er gitt i tabell 3.5.8. I optimalisert praktisk drift kan økningen bli større enn dette.

Slamseparasjon ved lamellsedimentering

Lamellsedimenteringsbassenger skal dimensjoneres for overflatebelastning på projisert flate, $v_p = Q/A_p$, der A_p er lik arealet i horisontalprojeksjonen av lamellenes samlede effektive areal (se tabell 3.5.9).

Tabell 3.5.9. Anbefalte dimensjoneringsdata for lamellsedimentering ved biofilmanlegg m/felling (MBBR, rislefilter og biorotor m/felling).

Overflatebelastning på projisert flate, v_p ($m^3/m^2 \cdot h$)	**Q_{dim}**	**$Q_{maksdim}$**
Ved separasjon av biofilmslam uten felling	0,4	0,6
Ved separasjon av biofilmslam med polymer-koagulering	0,5	0,8
Ved separasjon av biofilmslam etter felling (m/Al,Fe)	0,6	1,0
Ved separasjon av biofilmslam etter felling (m/Al,Fe) og polymer-koagulering	0,8	1,2

Slamseparasjon med flotasjon

Det finnes flere patenterte flotasjonsreaktorer levert av ulike leverandører med lameller, fordelingsbunner etc innebygget (f.eks. AquaDAF, Speedflo, etc.). For dimensjonering av slike spesifikke separasjonsreaktorer henvises det til de aktuelle leverandørene.

Dersom man baserer seg på egen utforming og dimensjonering, bør flotasjonsbassenget dimensjoneres med hensyn på overflatebelastning og oppholdstid, som angitt i tabell 3.5.10.

Tabell 3.5.10. Anbefalte dimensjoneringsdata for flotasjonsbasseng ved biofilmanlegg m/felling (MBBR, rislefilter og biorotor m/felling).

Anvendelse	**Overflatebelastning[1,2,3] $m^3/m^2 \cdot h$**	
	Q_{dim}	$Q_{maksdim}$
Separasjon av biofilmslam uten kjemisk felling (eller koagulering med polymer)	5	8
Separasjon av biofilmslam uten kjemisk felling , men med koagulering med polymer	5,5	10
Separasjon av biofilmslam med kjemisk felling (m/Al/Fe)	6	11
Separasjon av biofilmslam med kjemisk felling og med polymer tilsatt som hjelpeflokkulant	7	12

[1] *Basert på innkommende vann*

[2] *Forutsetter vanndyp > 2 m*

[3] *Basert på effektivt areal, dvs ikke inkludert arealet av dispergeringssonen*

Følgende veiledning gjelder for dimensjonering av dispergeringssystemet ved separasjon av kjemisk slam:

Trykk	: 400–600 kPa (4-6 bar)
Luftmetningsgrad	: 80-90 %
Dispersjonsvannmengde (% av $Q_{maksdim}$)	: 10-25 % (avh. av SS_{inn} og luftmetningsgrad)

Den løste luftmengden er direkte proporsjonal med produktet av dispersjonsvannmengde og trykk. Ved bruk av nedre grense for trykk skal øvre grense for dispersjons-vannmengde benyttes og omvendt.

Dispergeringsanlegg skal ha en minimumskapasitet tilsvarende 10 % av $Q_{maksdim}$ ved et trykk på 600 kPa.

Andre slamseparasjonsmetoder

Det finnes flere patenterte separasjonsreaktorer med integrerte flokkuleringsbasseng og med lameller, ledevegger eller lignende innebygget (Densadeg, Actiflo etc,). Disse dimensjoneres av leverandøren for vesentlig høyere overflatebelastning (30-70 m/h beregnet på vannoverflaten). For dimensjonering av slike spesifikke separasjonsreaktorer henvises det til de aktuelle leverandørene

Konstruktiv utforming

MBBR-anlegg skal være utstyrt med luftesystem (i aerobe reaktorer), omrørere (i anoksiske og anaerobe reaktorer), siler mellom hver reaktor og bæremedium for biofilm.

Luftesystem

Luftesystemet kan være utformet med middels grov/fin bobleluftere (som er det vanligste) eller finluftere. Om middels grov/fin boble luftere benyttes basert på perforerte rør (hulldiameter: 4 mm), bør følgende dimensjonerende (maksimale) lufthastigheter benyttes:

- Tilførsels rør: 16 m/s
- Fordelings rør (til manifold): 14 m/s
- Diffusor rør: 10 m/s

Antall diffusorhull bør tilpasses slik at gjennomstrømningen i hvert hull er 0,5 m^3/h -1,3 m^3/h. Luftesystemet skal utformes slik at det blir god omrøring i reaktoren f.eks. ved å ha noe færre diffusorhull i starten av reaktoren (20 % i første fjerdedel) og noe flere i slutten (30 % i siste fjerdedel).

Omrørere

Omrørere skal være slik utformet at de gir fullstendig omrøring i reaktoren, at de ikke påvirker biofilm bærerne negativt og at de beveger biofilmbærerne bort fra utløpssilene i reaktoren.

For dimensjonering kan man regne med følgende effektbehov for omrøring i reaktoren:
- Dykkede omrørere (m/skråstilt aksling): >20 W/m^3 (maks periferihastighet: 6 m/s)
- Omrørere med vertikal aksling: >15 W/m^3 (maks periferihastighet: 4 m/s)

I praksis kan effektbehovet bli mindre enn dette – avhengig av type omrører, basseng dyp og bassengutforming. Antall omrørere bestemmes ut fra effektbehovet til omrøring i reaktoren (W/m^3) og den aktuelle omrørerens nominelle effekt (W)

Reaktorsiler

Reaktorsiler kan ha ulike utforminger. Det vanligste er flate siler i omrørte (anoksiske og anaerobe) reaktorer og sylindriske siler i aerobe reaktorer. For å forhindre akkumulering av biofilm bærere mot silflaten, bør siler utstyres med sil-luftere plassert under silen. Luftmengden gjennom sil-lufteren bør være 0,1–0,2 ganger vannmengden gjennom silen

Siler skal ha et lysåpningsareal som minst utgjør 40 % av reaktorens gjennomstrømningsareal. Hulldiameteren i silens perforering skal ikke være større enn 60 % av biofilmbærerens største diameter.

Gjennomstrømningshastigheten i reaktoren (mot silen) bør ikke være større enn:

- Omrørte (anoksiske/anaerobe) reaktorer (flate siler): 30 m/h
- Luftede (aerobe) reaktorer (sylindriske siler): 35 m/h

Slamseparasjonsbassenger

For konstruktiv utforming av slamseparasjonsbassenger, se under pkt 3.4.4

Ressursforbruk

Energiforbruk

Energi forbrukes til lufting og omrøring i MBBR-anlegg. Man kan ved dimensjonering ta utgangspunkt i følgende verdier for energiforbruk til bioreaktoren (energiforbruk til slamseparasjonen samt oppvarming og ventilasjon kommer i tillegg, se pkt. 3.3.3):

- MBBR i sekundærrenseanlegg (anlegg for fjerning av BOF_5)
    - Effektforbruk til lufting: 7,5-10 W/m$^3_{\text{reaktorvolum}}$
    - Totalt energiforbruk (til lufting): 0,08 – 0,10 kWh/m$^3_{\text{vann behandlet}}$

- MBBR i tertiærrenseanlegg (anlegg for fjerning av nitrogen ved kombinert for- og etterdenitrifikasjon)
    - Effektforbruk til lufting: 25-35 W/m$^3_{\text{reaktorvolum}}$
    - Effektforbruk til omrøring: 10-15 W/m$^3_{\text{reaktorvolum}}$
    - Totalt energiforbruk (til omrøring, lufting og returpumping): 0,2 – 0,3 kWh/m$^3_{\text{vann behandlet}}$

Kjemikalieforbruk

Kjemikalieforbruket til fellingsdelen av MBBR-anlegg med felling er om lag det samme som ved kjemiske fellingsanlegg – se pkt. 3.4.5

Kompakte sedimenteringsbasseng, som benytter mikrosand som ballast i fnokkdannelsen i kjemiske rensetrinn, har et forbruk av mikrosand, som kan anslås til 3-5 g/m$^3_{\text{vann}}$.

IVL Svenska Miljöinstitutet AB // Box 210 60 // 100 31 Stockholm
Tel 010-788 65 00 // www.ivl.se