Effectiveness of non-fertilized buffer strips in the Netherlands

Final report of a combined field, model and cost-effectiveness study

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Effectiveness
Buffer strips

Effectiveness of non-fertilized buffer strips in the Netherlands

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Effectiveness of buffer strips publication series

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10. Noij, I.G.A.M., M. Heinen and P. Groenendijk, 2012. Effectiveness of non-fertilized buffer strips in the Netherlands. Final report of a combined field, model and cost-effectiveness study. Alterra-report 2290, Wageningen.

Abstract

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This integrated field and model study explains low effectiveness of non-fertilized 5 m wide harvested grass buffer strips (BS) to reduce nutrient loads from agricultural lowland fields to ditches. Buffer strip effectiveness (BSE) was defined as the relative nutrient load reduction, compared to a normally fertilized reference strip (REF). We collected nutrient loads from paired BS and REF strips in separate reservoirs in the ditch for three or four years at five field sites characteristic for the hydrogeology of the Netherlands. No statistically significant BSE was found at three out of five sites. A statistically significant BSE of 10-15% was found for N at the peat soil site and of 57-61% for P at the site with pure shallow flow. Dynamic 2D modelling shows BSE for N slightly increases after four years, the increase of BSE for P depends on the rate of net P withdrawal from the BS by the harvested grass and on the buffer capacity of the soil. BSE increases with BS width or ditch density. A steady-state model predicts BSE for N between 7 and 22% for sandy soils, and between 14 and 25% for peat soils. Highest BSE for N is expected on fields with high, but not pure shallow flow. High BSE for P is expected with high surface runoff or shallow flow, and with high original P status of the soil (i.e. P leaking soils). Buffer strips are ineffective on pipe drained fields (clay soils). According to our cost analysis BS may be cost-effective under specific circumstances, but more cost effective alternatives exist to reduce nutrient loads.

Keywords: agriculture buffer strip, cost analysis, effectiveness,hydrogeology, leaching, lowland, nitrogen, Phosphorus, surface water

The photograph on the cover was taken in Winterswijk before the extension of the reservoirs from 5 to 12.5 m.

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Wageningen, April 2012

Contents

Preface and acknowledgement

This research was funded by the Dutch Ministry of Economy, Agriculture and Innovation (EL&I), research project BO-12.12-002-019: 'Effectiveness of Buffer Strips in the Netherlands', and the Dutch Ministry of Infrastructure and Environment (I&M).

In 2005 both Dutch Ministries asked Alterra to prepare a project proposal on the assessment of the effectiveness of non-fertilized buffer strips in the Netherlands. Three workshops were held in 2005 in which several approaches of field work were discussed together with financial implications. After the third workshop consensus was achieved on a project consisting of three work packages: a field experiment, a modelling study and a cost-effectiveness study. The workshops were attended by representatives of the Ministries of EL&I and I&M, National institute for waste water treatment and inland water management (RIZA, later WD), Netherlands Environmental Assessment Agency (MNP, now PBL), University of Utrecht and the three Wageningen UR institutes Alterra, Plant Research International and Applied Plant Research.

We acknowledge Edo Biewinga (EL&I) and Douwe Jonkers (I&M) for initiating the project and the other members of the supervision committee for their contribution: Nancy Meijers, Maartje Oonk and Piet Soons (EL&I), Lukas Florijn, Paul Boers and Olga Clevering (I&M), Mariet Hefting (University of Utrecht), Jaap Willems (Netherlands Environment Assessment Agency) and Oene Oenema (Alterra, Wageningen UR). We thank the members of the committee for their critical comments, especially the scientific members: Olga Clevering, Mariet Hefting, Jaap Willems and Oene Oenema. Marcel Bakker and Rogier Westerhoff of TNO Netherlands Geographical Survey (GPR and EM31 measurements) and our colleagues Jan van Bakel, Harry Massop, Han te Beest and Reind Visschers were involved in the selection of locations, Arie Van Kekem also during the rest of the project. Han te Beest, Jan van Kleef, Antonie van den Toorn, Meint Veninga and Gerben Bakker were responsible for the construction and maintenance of the field equipment. We thank the Chemical and Biological Soil Laboratory (Alterra, Wageningen UR) for doing all the chemical analyses. Hanneke Heesmans was responsible for the data handling, screening and storage, and Jac Thissen (Biometris, Plant Research International) did the statistical analyses. Jan Willem van Groenigen and Eduard Hummelink carried out the deuterium tracer experiment. The deuterium contents were analysed with laser-absorption spectroscopy at the Davis Stable Isotope Facility of the University of California (USA). Falentijn Assinck did the modelling work for the Zegveld location. Special thanks are due to the farmers of the experimental locations for their cooperation: Huinink and Ribbers (Beltrum), Van Laarhoven (Loon op Zand), Hoitink (Winterswijk), and the heads of the two experimental stations Karel van Houwelingen, Animal Sciences Group, Wageningen UR (Zegveld) and Pierre Bakker, Plant Sciences Group, Wageningen UR (Lelystad). Jan Visscher (ASG Wageningen UR) coordinated crop sampling.

In September 2006 the project was reviewed by four international colleagues. We greatly acknowledge their input: Pascal Boeckx (University of Ghent, Belgium), Jane Hawkins (BBSRC Institute of Grassland and Environmental Research, United Kingdom), Brain Kronvang (National Environmental Research Institute, Silkeborg, Denmark) and Martyn Silgram (ADAS Wolverhampton, United Kingdom). Based on their comments the project was extended with additional treatment replicates at two locations, extra groundwater samples and an extra year.

Summary

Abstract

We studied the effectiveness of non-fertilized 5 m wide harvested grass buffer strips (BS) to reduce nutrient loads from agricultural lowland fields to the nearest surface water course. Buffer strip effectiveness (BSE) was defined as the relative nutrient load reduction by a BS, compared with a normally managed reference strip (REF). We measured BSE for three or four years at five characteristic field sites for the hydrogeology of the Netherlands by collecting nutrient loads from paired treatments strips (BS and REF) in separate reservoirs in the ditch. A dynamic 2D model was built to study the development of BSE in time and the effect of BS width. Another steady state model was developed to study the expected range of BSE for N in the Netherlands. For P we identified the key factors that determine BSE by interpretation of the field results. The costs and effectiveness of BS and alternative measures were calculated to compare the cost effectiveness.

No statistically significant BSE was found in our field experiments at three out of five sites. For N we found a low statistically significant BSE of 10-15% at the peat soil site. For P we found a high, statistically significant BSE of 57-61% at the site with pure shallow flow. BSE was low at most sites, because flow paths ran relatively deep, thus bypassing the top layers of the BS, and sometimes even the ditch with the reservoirs. Low BSE for nitrate could further be attributed to denitrification outside the BS due to relatively high groundwater levels, a process that also occurs in a situation without BS. According to our dynamic model, BSE for N will only slightly increase after the four years experimental period. The development of BSE for P depends on the rate of net P withdrawal from the BS and the buffer capacity[1] of the soil. For both nutrients BSE increases with BS width or lower distance between ditches (i.e. higher ditch density). Based on the variation in ditch density, our steady state model predicts BSE for N between 7 and 22% for sandy soils, and between 14 and 25% for peat soils. Highest BSE for N is expected on fields with relatively shallow, but not pure shallow flow. Buffer strips are ineffective on pipe drained fields (clay soils). The high BSE for P on the shallow flow site indicates that BS could be relevant for 'P-leaking soils'. BSE for P increases with the amount of surface runoff or shallow flow, and with original P status of the soil. According to the cost analysis BS may be cost-effective for specific circumstances, but more cost-effective alternatives exist to reduce nutrient loads.

[1] The term buffer capacity may be confusing in the context of this study. Buffer capacity is a soil chemical characteristic for the resistance of a soil to chemical change, defined for P as the ratio between P bounded to the solid phase of the soil and the P concentration in solution at equilibrium.

Rationale

At the beginning of our study limited data were available on BSE in plain deeply permeable lowlands, like the major part of the Netherlands. Such lowlands have a distinct hydrogeology with relatively deep groundwater flow paths (Figure 1), low surface runoff, high groundwater levels, many tile drained fields and abundant ditches. Deep flow paths loaded with nutrients by pass the top layers of the BS, potentially reducing their effectiveness. This also occurs in case of drain pipes. Low surface runoff would implicate little sediment transport with associated particulate nutrients, thus reducing interception of nutrients by a BS. High groundwater levels would stimulate denitrification in a reference situation without BS, which according to the definition reduces BSE. Additionally, installing BS along the abundant water courses in the Netherlands would imply a relatively high claim on agricultural land with high costs.

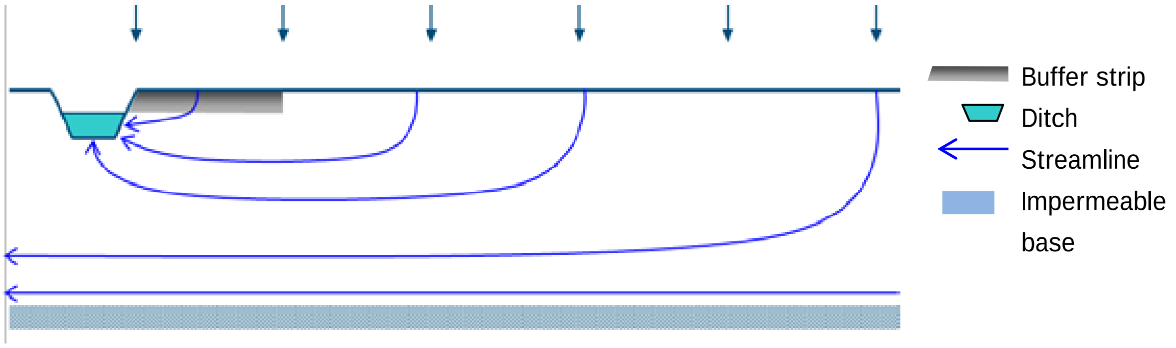


Figure 1
Predominant hydrogeological situation in the Netherlands with flow paths by passing the BS

Definitions

The buffer strip (BS) considered best suited for the Netherlands was a 5 m wide non-fertilized field border with harvested grass to remove nutrients. On grassland the BS was allowed to be grazed. Buffer strip effectiveness (BSE) was defined as the relative reduction of the nutrient load by an non-fertilized BS, as compared with a nearby reference strip (REF) that was normally cultivated and fertilized like the rest of the field. The reference for the study was based on Good Agricultural Practice, including a small obligatory uncultivated strip of 25 cm on grassland and 50 cm on maize. This implies that the potentially beneficial effect of BS in cases where a farmer or contractor does not adhere to GAP, has not been taken into account. For instance, in case of fertilizer spreading and tillage too close to the ditch, a BS would reduce the direct impact of fertiliser or ditch bank erosion. We only considered the nutrient load to the nearest water course, mostly a field ditch, and no potential BS effects on loads to water courses of higher order further away.

Three effects

We distinguished three conceptual BS effects, although we did not measure or model these separately. The fertilizer effect is the trivial effect of area weighted fertilizer rate reduction: a 5 m wide BS on a 100 m field implies 5% reduction in fertilizer rate, and therefore a lower nutrient load. It is the same effect that can be expected from a 5% fertilizer rate reduction on the whole field, and does not motivate installing a BS. Hence, a BS can only be considered effective if BSE exceeds the fertilizer effect (>5% in the example above). The two other effects, travel time effect and interception effect, are due to the specific position adjacent to the ditch, and might motivate installing a BS. Water with nutrients following the shorter flow paths near the ditch (Figure 1 left) travel less time before reaching the ditch, compared to water starting further away (Figure 1 right). As there is less time for nutrient retention, these shorter flow paths contribute more to nutrient load. The travel time effect of a BS is caused by specifically affecting the shorter flow paths starting near the ditch. The interception effect is caused by retention of nutrients on or in the soil of the BS, originating from surface runoff and flow paths starting outside the adjacent strip.

Three goals

The first and major goal of this research was to collect experimental evidence for BSE on characteristic sites for the Netherlands. The second objective was to model BSE for longer periods of time, and for other site conditions in the Netherlands. The model should allow us to assess the variability of BSE in space and time. The third goal was to compare the cost-effectiveness of BS with alternative measures to reduce nutrient loads from agriculture to surface water.

Field study

The experimental research was conducted for three or four years at five sites that were characteristic for the five major hydrogeological classes in the Netherlands. Each site had a different soil type and land use (three grassland and two arable land with maize). The paired treatments BS and REF were replicated three times at two sites to enable statistical analysis. Nutrient loads from BS and REF were measured with separate reservoirs built in the ditch. Load reduction by the BS was small, both for N and P. At three sites no statistically significant effect of BS was found. In Zegveld (grassland on peat soil), a small statistically significant BSE of 10-15% was found, only for N. In Winterswijk (sloping grassland on very shallow soil) a statistically significant BSE of 57-61% was measured, only for P. Groundwater measurements proved insufficient to assess BSE.

Why low BSE?

Low BSE was explained by a series of site-specific hydrogeological factors, such as a low shallow flow (Beltrum, Loon op Zand, Lelystad), high downward seepage (Beltrum, Loon op Zand), low residence time in the BS (Winterswijk, Lelystad), pipe drainage (Lelystad), and surface runoff towards the centre of the field (Zegveld), all causing nutrient transport bypassing the BS. Low BSE for nitrate could further be explained by denitrification outside the BS, as deduced from upper groundwater measurements. Denitrification in the remainder of the field and the adjacent ditch bank also occurred at REF, which by definition decreased BSE. Low BSE for P was due to the absence of surface runoff and shallow flow at most sites, except in Zegveld, where high buffer capacity of the peat soil caused low BSE for P. The high BSE for P measured in Winterswijk was caused by the rare site specific combination of pure shallow groundwater flow, high original P status and low buffer capacity of the soil. Winterswijk belongs to a hydrogeological class comprising 2.5% of the country.

Effect of time

A dynamic 2D nutrient model was developed to describe water and nutrient transport in a transect perpendicular to the ditch, and to calculate the effect of time and BS width on BSE. Model input was taken from the two field sites Beltrum (maize on deep sandy soil; only N) and Zegveld (grassland on peat soil; both N and P). The remainder of model parameters was found in literature and by calibration with field data. According to the model, BSE for N will slightly increase after the four years experimental period. It will take another 10-15 years to reach the 2-4% higher final or maximum BSE for N. In Zegveld, BSE for P would increase from 2 to 3.5% over the next 50 years, but even then steady state is not reached, due to the very high buffer capacity of the peat soil. The necessary time span to reach maximum BSE for P will be lower on mineral soils with lower soil buffer capacity and also with a higher rate of net P withdrawal, but the model was not applied yet to such situations. Nevertheless, other P mining experiments have shown it takes decades for such a measure to become fully effective.

Effect of width

An analytical steady state model was developed for studying the expected range of BSE for N in the Netherlands. The model describes BSE for N as a function of (1) BS width and distance between ditches, (2) depth of active layer and aquifer, and (3) decomposition rates in active layer and below. The model was calibrated with the long term (near steady state) results of the dynamic 2D nutrient model. According to both models BSE increases with BS width or lower distance between ditches. With a fixed BS width (5 m), BSE is dependent on the distance between ditches (i.e. ditch density). In case of wider BS it takes more time to reach maximum BSE, both for N and P.

Upscaling BSE

The low BSE for N found at the two sandy sites with deep (Beltrum) and pure shallow flow (Winterswijk) correspond with the observation of Hill (1996) that there is an optimal flow depth range for BSE for N. However, we could not yet confirm the suggested optimal flow depth range (1 to 4 m bss), because BSE for N was also low at Loon op Zand, the site where an intermediate flow depth was expected, but did not occur. Anyhow, there are no simple indicators to identify fields with intermediate flow, where higher BSE for N might be found. We delineated an area of about 10% of the agricultural land, where such situations might be found.

According to our model analysis the expected range of BSE for N is mainly determined by the hydrogeological factors ditch density, aquifer depth and amount of seepage flow. Based on the spatial variation of the most important variable ditch density, our analytical model calculates final BSE for N between 7 and 22% for sandy soils, and between 14 and 25% for peat soils. Involving other spatial variables would require a more extensive hydrogeological study, because these variables are spatially intertwined. Buffer strips are ineffective on pipe drained fields (most clay soils).

Shallow flow or surface runoff is a precondition for effectively reducing P loads with BS. Although fields with only shallow flow like in Winterswijk are rare in the Netherlands, the high BSE for P observed there, indicates that BS could be relevant for 'P-leaking soils'. Final BSE for P is expected to be positively correlated with the amount of shallow flow or surface runoff and with original P status of the soil. Maximum BSE for P will be reached faster with higher net P withdrawal from the BS and with lower buffer capacity of the soil. Although not observed in our experiments, surface runoff does also occur on plain fields, especially from very local pools during the second half of the winter period, caused by high groundwater level or stagnating soil layers. This process is under study and cannot yet be quantified for the Netherlands. Alternative more cost effective measures for BS are available to mitigate surface runoff.

Cost analysis

The costs and effects on nutrient loads of the alternative source and hydrological measures were calculated using a model chain, including arable and dairy farm models, a hydrological model and the nutrient model also used in the dynamic 2D model. Costs and effects of constructed wetlands and wetland buffer strips were based on literature data. Buffer strips may be cost-effective for specific circumstances, but there are more cost-effective alternatives. Buffer strips are cost-effective if the absolute load reduction is more than several kilos of N or more than several tenths of kilos of P per hectare per year. On arable farms with clay soil the effectiveness of BS must be more than 20% for N to be cost-effective, while on other farms more than 10% may be enough. Hardly any costs of BS were calculated on dairy farms with clay or peat soil that are self-sufficient for roughage. Therefore, BS are cost-effective on such farms, even with a low BSE for N, although more effective measures are available. On 'P-leaking soils' with sufficient shallow flow, BS are cost-effective and more cost-effective than P mining of the whole field, but more cost-effective alternative measures are available.

Alternative measures to protect surface water against nutrient loads comprise extra source measures compared to existing manure policy, hydrological measures such as blocking surface runoff and installing pipe drainage, and constructed wetlands or wetland buffers. As surface runoff does not take place along the entire length of the ditch, it would be more cost-effective to focus measures on the spots where it does. Alternative measures to reduce surface runoff could be a well-designed narrow trench or barrier, pipe drainage, soil surface levelling or soil improvement. Although none of the studied alternatives is cost-effective everywhere, at least one of the alternatives is always more cost-effective than BS. Apparently, cost-effective reduction of nutrient loads to surface water requires tailor made measures.

We only evaluated the benefits of BS for reducing nutrient loads from agriculture to surface water. However, multifunctional BS can have additional beneficial effects, such as reducing pesticide loads, stabilizing ditch banks, reducing soil erosion, reducing costs for ditch maintenance such as dredging, and increasing biodiversity, ecological connectivity and functional agro-biodiversity (reducing pest risk). Wetland buffer strips may further be designed to reduce peak discharges and facilitate higher surface water levels to mitigate desiccation. The cost effectiveness of a multifunctional BS is better, either because more benefits could be attributed to the measure, or the costs would be divided over more goals.

Publication

An 'Effectiveness buffer strips publication series' was issued (see Page 5), including the report on cost effectiveness, and concluding with this final report. The results of the field study have been and are being published in peer reviewed journals.

Heinen, M., I.G.A.M. Noij, H.I.M. Heesmans, J.W. van Groenigen, P. Groenendijk and J.T.N.M. Thissen, 2012. A novel method to determine buffer strip effectiveness on deep soils. J. Environ. Qual. 41(2): 334-347. doi:10.2134/jeq2010.0452.

Noij, I.G.A.M., M. Heinen, H.I.M. Heesmans, J.T.N.M. Thissen and P. Groenendijk, 2012. Effectiveness of non-fertilized buffer strips to reduce nitrogen loads from agricultural lowland to surface waters. J. Env. Qual. 41(2): 322-333. doi:10.2134/jeq2010.0545

Noij, I.G.A.M., M. Heinen, H.I.M. Heesmans, J.T.N.M. Thissen and P. Groenendijk, in prep. Effectiveness of non-fertilized buffer strips to reduce phosphorus loads from agricultural lowland to surface waters. Submitted tot Soil Use and Management.

Further peer reviewed publication is foreseen on the modelling and cost effectiveness study.

Samenvatting

Abstract

We hebben de effectiviteit bestudeerd van 5 meter brede onbemeste bufferstroken (BS) met gemaaid gras om de belasting van het naastgelegen oppervlaktewater met nutriënten vanuit landbouwpercelen te verminderen. De effectiviteit van bufferstroken (BSE) is gedefinieerd als de relatieve vermindering van de nutriëntenbelasting door een BS ten opzichte van een normaal beheerde referentiestrook (REF). We hebben BSE drie of vier jaar lang gemeten op vijf percelen die qua geohydrologie kenmerkend zijn voor Nederland, door de nutriëntenvrachten van gepaarde behandelstroken (BS en REF) op te vangen in aparte bakken in de sloot. Er is een dynamisch 2D-model gebouwd om de ontwikkeling van BSE in de tijd, en het effect van de breedte van de BS te bestuderen. Daarnaast is een evenwichtsmodel ontwikkeld om de verwachte bandbreedte van BSE voor N in Nederland te bestuderen. Voor P hebben we vastgesteld welke factoren het meest van invloed zijn op BSE door de resultaten van het veldonderzoek te interpreteren. Om de kosteneffectiviteit te vergelijken, zijn de kosten en effectiviteit van BS en alternatieve maatregelen berekend.

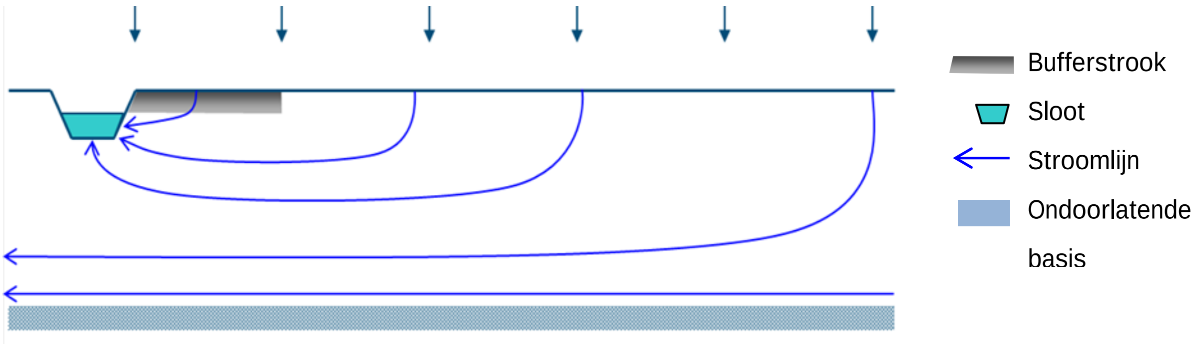
Op drie van de vijf percelen is geen statistisch significante BSE gevonden bij onze veld-experimenten. Voor N hebben we een lage statistisch significante BSE gevonden van 10-15% op het perceel met veengrond. Voor P hebben we een hoge statistisch significante BSE gevonden van 57-61% op het perceel met uitsluitend ondiepe afvoer. Op de meeste percelen was BSE laag omdat de relatief diepe stroombanen niet of nauwelijks contact maakten met de toplagen van de BS en soms zelfs onder de sloot met de bakken door liepen. Een lage BSE voor nitraat kon verder worden toegeschreven aan denitrificatie buiten de BS door een relatief hoge grondwater-spiegel, een proces dat zich ook voordoet op percelen zonder BS. Volgens ons dynamische model zal BSE voor N slechts licht toenemen na de periode van vier jaar. De ontwikkeling van BSE voor P is afhankelijk van de netto P-opname uit de BS en de buffercapaciteit[2] van de grond. Voor beide nutriënten neemt BSE toe naarmate de BS breder is of de afstand tussen de sloten kleiner (hogere slootdichtheid). Op basis van de variatie in slootdichtheid voorspelt ons evenwichtsmodel een BSE voor N van 7-22% voor zandgrond en 14-25% voor veengrond. De hoogste BSE voor N wordt verwacht op percelen met relatief veel, maar niet uitsluitend ondiepe afvoer. Bufferstroken hebben geen effect op percelen met buisdrainage (kleigrond). De hoge BSE voor P op het perceel

[2] De term buffercapaciteit kan in de context van deze studie wat verwarrend zijn. De buffercapaciteit is een chemische eigenschap van de bodem, de weerstand van de bodem tegen chemische verandering, voor P gedefinieerd als de evenwichtsverhouding tussen de hoeveelheid P gebonden aan de vaste fase en de P-concentratie in oplossing.

met uitsluitend ondiepe afvoer toont aan dat BS relevant kunnen zijn voor fosfaatlekkende gronden. BSE voor P neemt toe met de hoeveelheid afspoeling of ondiepe afvoer en met de oorspronkelijke P-toestand van de grond. Volgens de kostenanalyse kunnen BS in bepaalde situaties kosteneffectief zijn, maar bestaan er kosteneffectievere alternatieven om de nutriëntenbelasting te verlagen.

Aanleiding

Bij het begin van onze studie waren beperkt data beschikbaar over BSE op vlak, relatief diep doorlatend laagland zoals in het grootste deel van Nederland. Zulk laagland heeft een andere geohydrologie met relatief diepe grondwaterstroombanen (Figuur 2), een lage maaiveldafvoer, een hoge grondwaterspiegel, veel gedraineerde percelen en een hoge slootdichtheid. Diepe stroombanen met nutriënten stromen onder de toplagen van de BS door, waardoor BSE mogelijk afneemt. Dit doet zich ook voor in het geval van drainbuizen. Een lage maaiveldafvoer zou weinig sedimenttransport met nutriënten met zich meebrengen, waardoor er minder nutriënten kunnen worden onderschept door een BS. Een hoge grondwaterspiegel zou denitrificatie ook stimuleren in een referentiesituatie zonder BS, waardoor BSE per definitie minder wordt. Bovendien zou het installeren van BS langs de vele waterlopen in Nederland relatief veel beslag leggen op landbouwgrond, met hoge kosten als gevolg.



Figuur 2
Veel voorkomende geohydrologische situatie in Nederland met stroombanen die onder de BS door lopen

Definities

De bufferstroken (BS) die het meest geschikt werden geacht voor Nederland zijn 5 meter brede, onbemeste stroken met gemaaid gras om nutriënten aan de bodem te onttrekken. In het geval van grasland is beweiding in de BS toegestaan. De effectiviteit van bufferstroken (BSE) is gedefinieerd als de relatieve vermindering van de nutriëntenbelasting door een onbemeste BS ten opzichte van een nabij gelegen referentiestrook (REF) die op dezelfde manier wordt beteeld en bemest als de rest van het perceel. De referentie voor deze studie was gebaseerd op 'Goede LandbouwPraktijk' (GLP), inclusief de verplichte teeltvrije zone van 25 cm bij grasland en 50 cm bij maïs. Dit betekent dat geen rekening is gehouden met het mogelijk gunstige effect van BS wanneer een boer of loonwerker niet volgens GLP werkt. In het geval van bemesting of grondbewerking te dicht in de buurt van de sloot zou een BS de directe belasting met meststoffen of door erosie van de slootkant verminderen. We hebben alleen de nutriëntenbelasting meegenomen van de dichtstbijzijnde waterloop, meestal een sloot, en de mogelijke effecten van BS op de belasting van verder weg gelegen waterlopen van hogere orde buiten beschouwing gelaten.

Drie effecten

We hebben het totale effect van de BS opgedeeld in drie conceptuele effecten, hoewel we deze effecten niet afzonderlijk hebben gemeten of gemodelleerd. Het bemestingseffect is het triviale effect van de kleinere hoeveelheid meststof die wordt toegediend aan het perceel: een 5 m brede BS op een perceel van 100 m betekent 5% minder toediening van meststoffen. Ditzelfde effect mag ook worden verwacht van 5% minder toediening van meststoffen op het gehele perceel en is dus geen aanleiding om een BS aan te leggen. Een BS kan daarom alleen als effectief worden beschouwd als BSE het bemestingseffect overschrijdt (>5% in het bovenstaande voorbeeld). De twee andere effecten - het verblijftijdeffect en het onderscheppend effect - worden veroorzaakt door de specifieke ligging van de BS naast de sloot en kunnen wel een reden zijn om een BS aan te leggen. Water met nutriënten dat de kortere stroombanen vlakbij de sloot (figuur 2 links) volgt, is minder lang onderweg naar de sloot dan het water dat van de rest van het perceel komt (figuur 2 rechts). Doordat er minder tijd is om de nutriënten uit het water te halen, dragen deze kortere stroombanen relatief veel bij aan de nutriëntenbelasting. Het verblijftijdeffect van een BS wordt veroorzaakt doordat BS juist deze kortste stroombanen vlakbij de sloot beïnvloeden. Het onderscheppend effect wordt veroorzaakt doordat nutriënten uit maaiveldafvoer en stroombanen die buiten de BS beginnen op of in de grond van de BS achterblijven.

Drie doelen

Het eerste en belangrijkste doel van dit onderzoek was om experimenteel bewijs te vinden voor BSE op percelen die kenmerkend zijn voor Nederland. Het tweede doel was om BSE voor langere tijdsperioden en voor andere perceelomstandigheden in Nederland te modelleren. Met behulp van het ontwikkelde model moest de bandbreedte van BSE in Nederland in beeld worden gebracht. Het derde doel was om de kosteneffectiviteit van BS te vergelijken met alternatieve maatregelen om de nutriëntenbelasting van oppervlaktewater vanuit landbouwpercelen te verlagen.

Veldonderzoek

Het experimentele onderzoek is drie of vier jaar lang uitgevoerd op vijf percelen die kenmerkend zijn voor de vijf meest voorkomende geohydrologische situaties in Nederland. Elk perceel had een ander bodemtype en landgebruik (drie grasland en twee akkerbouw met maïs). De gepaarde stroken (BS en REF) werden op twee locaties drie keer herhaald om een statistische analyse mogelijk te maken. De nutriëntenbelasting van de BS en de REF werd gemeten met aparte bakken in de sloot. De vermindering van de belasting door de BS was klein, zowel voor N als voor P. Op drie percelen werd geen statistisch significant effect van de BS geconstateerd. In Zegveld (grasland op veengrond) werd een lage statistisch significante BSE gevonden van 10-15%, alleen voor N. In Winterswijk (hellend grasland op zeer ondiep ondoorlatende bodem) werd een statistisch significante BSE gemeten van 57-61%, alleen voor P. Grondwatermetingen waren onvoldoende om BSE te bepalen.

Waarom lage BSE? Een lage BSE werd verklaard door een aantal locatie specifieke geohydrologische factoren die er allemaal voor zorgen dat het nutriëntentransport niet door de BS voert, zoals een beperkte maaiveldafvoer (Beltrum, Loon op Zand, Lelystad), een hoge mate van wegzijging (Beltrum, Loon op Zand), een korte verblijftijd in de BS (Winterswijk, Lelystad), buisdrainage (Lelystad) en maaiveldafvoer richting het midden van het perceel (Zegveld). Een lage BSE voor nitraat kon verder worden verklaard door denitrificatie buiten de BS, zoals afgeleid uit metingen van het bovenste grondwater. Denitrificatie in de rest van het perceel en in de slootkant trad ook op bij de REF, waardoor BSE per definitie daalde. Een lage BSE voor P werd veroorzaakt door de afwezigheid van maaiveldafvoer op de meeste locaties, behalve in Zegveld, waar een hoge buffercapaciteit van de veenbodem een lage BSE voor P veroorzaakte. De hoge BSE voor P die werd gemeten in Winterswijk werd veroorzaakt door de zeldzame locatie-specifieke combinatie van uitsluitend ondiepe grondwaterafvoer en een hoge oorspronkelijke P-toestand en lage buffercapaciteit van de bodem. Winterswijk behoort tot een geohydrologische situatie die 2,5% van het land beslaat.

Effect van tijd Er is een dynamisch 2D-model ontwikkeld om het water- en nutriëntentransport te beschrijven in een dwarsdoorsnede loodrecht op de sloot en om het effect van tijd en BS breedte op BSE te berekenen. De gegevens voor het model zijn afkomstig van twee percelen, één in Beltrum (maïs op diepe zandgrond; alleen N) en één in Zegveld (grasland op veengrond; zowel N als P). De overige modelparameters zijn afkomstig uit de literatuur of zijn geijkt met veldgegevens. Volgens het model zal BSE voor N licht toenemen na de periode van vier jaar. Het zal daarna nog zo'n 10-15 jaren duren voor een 2-4% hogere definitieve of maximale BSE voor N wordt bereikt. In Zegveld zou de BSE voor P in de komende 50 jaar toenemen van 2 tot 3,5%, maar zelfs dan is nog geen evenwicht bereikt, aangezien de buffercapaciteit van de veengrond zeer hoog is. De benodigde tijdspanne om een maximale BSE voor P te bereiken zal korter zijn op minerale gronden met een lagere buffercapaciteit en ook wanneer een hogere netto P-opname wordt gerealiseerd, maar het model is nog niet op dergelijke situaties toegepast. Ander veldonderzoek naar het uitmijnen van P heeft echter al aangetoond dat het tientallen jaren duurt voordat een dergelijke maatregel volledig effect bereikt.

Effect van breedte Er is een analytisch evenwichtsmodel ontwikkeld om de ruimtelijke variabiliteit van de definitieve BSE voor N te bestuderen. Het model beschrijft BSE voor N als een functie van (1) de breedte van de BS en de afstand tussen de sloten, (2) de diepte van de actieve laag en de aquifer, en (3) de afbraaksnelheid in de actieve laag en daaronder. Het model is geijkt aan de langetermijnresultaten (bijna evenwicht) van het dynamische 2D-nutriëntenmodel. Volgens beide modellen neemt BSE toe naarmate de BS breder is of de afstand tussen de sloten kleiner. Bij een vaste BS breedte (5 m) hangt BSE samen met de afstand tussen de sloten (slootdichtheid). Bij bredere BS duurt het zowel voor N als voor P langer om de maximale BSE te bereiken.

Opschalen van BSE

De lage BSE voor N die werd gevonden op de twee zandpercelen met diepe (Beltrum) en uitsluitend ondiepe grondwaterafvoer (Winterswijk) komt overeen met de waarneming van Hill (1996) dat er een optimale afvoerdiepte bestaat voor BSE voor N. We konden de gesuggereerde optimale afvoerdiepte (1 tot 4 m beneden maaiveld) echter niet bevestigen, omdat BSE voor N ook laag was in Loon op Zand, het perceel waar een tussenliggende afvoerdiepte werd verwacht, maar niet optrad. Hoe dan ook, er zijn geen eenvoudige indicatoren voor percelen met een tussenliggende afvoerdiepte, waar mogelijk een hogere BSE voor N kan worden gevonden. We konden een gebied afbakenen van circa 10% van de landbouwgrond waar dergelijke situaties mogelijk zouden kunnen optreden.

Volgens onze modelanalyse wordt de ruimtelijke variabiliteit van BSE voor N hoofdzakelijk bepaald door de geohydrologische factoren slootdichtheid, diepte van de aquifer en mate van wegzijging. Op basis van de belangrijkste ruimtelijke variabele slootdichtheid berekent ons analytische model een definitieve BSE voor N van 7-22% voor zandgrond en 14-25% voor veengrond. Om rekening te kunnen houden met de andere variabelen zou een uitgebreidere geohydrologische studie noodzakelijk zijn, omdat deze variabelen ruimtelijk zijn verstrengeld. Bufferstroken hebben geen effect op percelen met buisdrainage (de meeste kleigronden).

Maaiveldafvoer of ondiepe afvoer is een voorwaarde om de P-belasting effectief te kunnen verlagen met behulp van BS. Hoewel percelen met uitsluitend ondiepe afvoer, zoals in Winterswijk, zeldzaam zijn in Nederland, toont de daar gevonden hoge BSE voor P aan dat BS relevant kunnen zijn voor fosfaatlekkende gronden. De definitieve BSE voor P zal naar verwachting positief gecorreleerd zijn met de hoeveelheid maaiveldafvoer of ondiepe afvoer en met de oorspronkelijke P-toestand van de grond. De maximale BSE voor P zal sneller worden bereikt met een hogere netto P-opname uit de BS en in geval van een lagere buffercapaciteit van de bodem.

Hoewel dit tijdens onze experimenten niet is waargenomen, treedt maaiveldafvoer ook op bij vlakke percelen, vooral vanuit zeer plaatselijke waterplassen in de tweede helft van de winter. Dit wordt veroorzaakt door een hogere grondwaterspiegel of door stagnerende grondlagen. Dit proces wordt nader onderzocht en kan nog niet worden gekwantificeerd voor Nederland. Er zijn ook andere maatregelen om maaiveldafvoer te beperken.

Kostenanalyse

De kosten en de effecten op de nutriëntenbelasting van de alternatieve brongerichte en hydrologische maatregelen zijn berekend met een modellenketen inclusief een akkerbouw- of een melkveehouderijmodel, een hydrologisch model en het nutriëntenmodel dat ook is gebruikt voor het dynamische 2D-model. De kosten en effecten van vloeivelden en moerasbufferstroken zijn gebaseerd op gegevens uit de literatuur. Bufferstroken kunnen in specifieke omstandigheden kosteneffectief zijn, maar er bestaan alternatieven die kosteneffectiever zijn. Bufferstroken zijn kosteneffectief als de absolute vermindering van de belasting per hectare per jaar meer is dan een paar kilo's N of tienden kilo's P. Voor akkerbouwbedrijven met kleigrond moet de effectiviteit van BS voor N hoger liggen dan 20% om kosteneffectief te zijn, terwijl voor andere bedrijven een percentage van meer dan 10% voldoende kan zijn. Er is berekend dat BS voor melkveehouderijen met klei- of veengrond, die zelfvoorzienend zijn qua ruwvoer, zeer weinig kosten met zich meebrengen. Daarom zijn BS voor dergelijke bedrijven bijna altijd kosteneffectief, zelfs met een lage BSE voor N, hoewel effectievere maatregelen beschikbaar zijn. Op fosfaatlekkende gronden met voldoende ondiepe afvoer zijn BS kosteneffectief, en kosteneffectiever dan het uitmijnen van P op het hele perceel, maar er bestaan maatregelen die nog kosteneffectiever zijn.

Alternatieve maatregelen om het oppervlaktewater te beschermen tegen nutriëntenbelasting omvatten extra brongerichte maatregelen ten opzichte van het huidige mestbeleid, hydrologische maatregelen zoals het blokkeren van maaiveldafvoer en het toepassen van buisdrainage, en vloeivelden of moerasbufferstroken. Aangezien maaiveldafvoer niet over de hele lengte van de sloot plaatsvindt, zou het kosteneffectiever zijn om maatregelen toe te spitsen op de plekken waar dit proces wel optreedt. Alternatieve maatregelen om maaiveldafvoer te verminderen zijn bijvoorbeeld een goed ontworpen smalle greppel of barrière, buisdrainage, egalisatie of bodemverbetering. Hoewel geen van de bestudeerde alternatieven overal kosteneffectief is, is altijd een van de alternatieven kosteneffectiever dan BS. Blijkbaar vereist een kosteneffectieve vermindering van de belasting van het oppervlaktewater met nutriënten maatwerk voor elke locatie.

We hebben alleen de voordelen van BS geëvalueerd voor het verminderen van de oppervlaktewaterbelasting met nutriënten vanuit de landbouw. Multifunctionele BS kunnen echter extra voordelen bieden, zoals de vermindering van de belasting met pesticiden, de stabilisatie van slootkanten, het terugdringen van bodemerosie, het verlagen van kosten van slootonderhoud (baggeren), en het verhogen van biodiversiteit, het herstel van ecologische verbindingen en de toename van functionele agrobiodiversiteit (verlagen plaagdruk). Moerasbufferstroken kunnen zo worden ontworpen dat de piekafvoer wordt beperkt en een hogere waterstand mogelijk wordt gemaakt als maatregel tegen verdroging. De kosteneffectiviteit van multifunctionele BS wordt hoger naarmate er meer gunstige effecten zijn en de kosten over meer doelen kunnen worden verdeeld.

Publicaties

Er is een "Effectiveness of buffer strips publication series" uitgegeven (zie pagina 5) inclusief het rapport over kosteneffectiviteit en dit eindrapport. De resultaten van het veldonderzoek zijn en worden gepubliceerd in wetenchappelijke tijdschriften met peer review.

Heinen, M., I.G.A.M. Noij, H.I.M. Heesmans, J.W. van Groenigen, P. Groenendijk, en J.T.N.M. Thissen. 2012. A novel method to determine buffer strip effectiveness on deep soils. J. Environ. Qual. 41(2): 334-347. doi:10.2134/jeq2010.0452.

Noij, I.G.A.M., M. Heinen, H.I.M. Heesmans, J.T.N.M. Thissen, P. Groenendijk, 2012. Effectiveness of non-fertilized buffer strips to reduce nitrogen loads from agricultural lowland to surface waters. J. Env. Qual. 41(2): 322-333. doi:10.2134/jeq2010.0545

Noij, I.G.A.M., M. Heinen, H.I.M. Heesmans, J.T.N.M. Thissen, P. Groenendijk, in prep. Effectiveness of non-fertilized buffer strips to reduce phosphorus loads from agricultural lowland to surface waters. Submitted tot Soil Use and Management.

Verder is wetenchappelijke publictie voorzien over de modelstudie en de studie over kosteneffectiviteit.

1 Introduction

1.1 Political context

This study was initiated in 2004 in response to an agreement between the Netherlands and the European Union regarding the Third Dutch Action Program for the Nitrates Directive (2004-2009). The European Union (EU) had suggested the Netherlands to install non-fertilized buffer strips (BS) of at least 5 m wide along waterways in order to reduce nutrient loads from agricultural land to surface waters, like in other European countries. The Dutch Government had doubts about the cost-effectiveness of BS under Dutch conditions. On the other hand, it did not want to exclude the possibility of implementing this measure with an eye to the water quality targets specified in the EU Water Framework Directive. Therefore, it was agreed to undertake this large-scale study on the effectiveness of BS in the Netherlands, which lasted from October 2005 until the end of 2011. It was funded by the Dutch Ministry of Economy, Agriculture and Innovation (EL&I; previously LNV), and the Dutch Ministry of Infrastructure and Environment (I&M; previously VROM), both responsible for the implementation of the Nitrates Directive.

1.2 What type of buffer strips?

There are different sorts of BS, including wider natural riparian zones, constructed or natural wetland buffer strips, and 'dry' field border strips. Such BS may be left to develop naturally, or may be heavily managed with removal of vegetation and sediment in cases with erosion. For the Netherlands, non-fertilized dry narrow grass field borders were considered by far most relevant for agricultural land. The grass from the BS should be harvested to remove nutrients from the BS. On grassland fields, cattle grazing should be allowed for practical reasons, although this would to some extent compensate for the nutrient removal. Our study was, therefore, restricted to 5 m wide dry BS with grass. We defined BS effectiveness (BSE) as relative nutrient load reduction.

1.3 Scientific context

Non-fertilized BS are a widely recognized mitigation option to reduce N and P transport from agricultural fields to surface waters, as appears from scientific literature reviews on BSE from various countries (e.g., Barling and Moore, 1994, Australia; Dorioz et al., 2006, France; Dosskey, 2002, USA; Mayer et al., 2005, 2007, USA; Muscutt et al., 1993, UK; Parkyn, 2004, New Zealand; Polyakov et al., 2005, USA & Canada; Wenger, 1999, USA). Slope, landscape and hydrogeology are key factors governing BSE (Burt et al., 2002; Dorioz et al., 2006; Hill, 1996; Hoffmann et al., 2006, 2009; Mayer et al., 2005, 2007; Puckett 2004; Ranalli and Macalady, 2010; Rassam et al., 2008; Sabater et al., 2003; Vidon and Hill, 2004; White and Arnold, 2009). However, there is no experimental evidence for BSE on well drained plain deltas with deeply permeable soil, such as the major part of the Netherlands. Van Beek et al. (2007) studied the effect of a grass BS on upper groundwater quality below a sandy soil in the Netherlands. However, groundwater quality as such, cannot be used for establishing BSE, as will be shown in Chapter 2.Two other studies from the Netherlands that include experimental data (Hefting, 2003; Hefting et al., 2003, 2006; Hendriks et al., 1996) and focus on surface water, were both restricted to a relatively small Eastern area of the Netherlands with slope and sub soil of low permeability, which is exceptional for the Netherlands. Under such circumstances much runoff and superficial discharge may be expected from nearby agricultural land containing nitrogen (N) and phosphorus (P), either in soil particles (especially P) or in soluble form (especially N). Soil particles in run-off can be filtered out by BS vegetation, while soluble N and P may be retained by net withdrawal in the vegetation, and by denitrification (N) or adsorption (P) in the soil of the BS. Under the prevailing 'Dutch conditions', without slope and with deeply permeable or pipe drained soils, most discharge by-passes below the active topsoil layer of the BS, which is expected to reduce BSE. Borin and Bigon (2002) and Borin et al*.* (2005) found high BSE for N for a traditional combined grass (5 m) + tree (1 m) buffer strip in the Po valley in Italy, a hydrogeological situation which is to some extent comparable with the Dutch delta. However, they assessed BSE without a reference treatment, which according to Dosskey (2002) and Chapter 2 likely leads to overestimation of BSE. Moreover, BSE was partly attributed to the effect of deep tree roots.

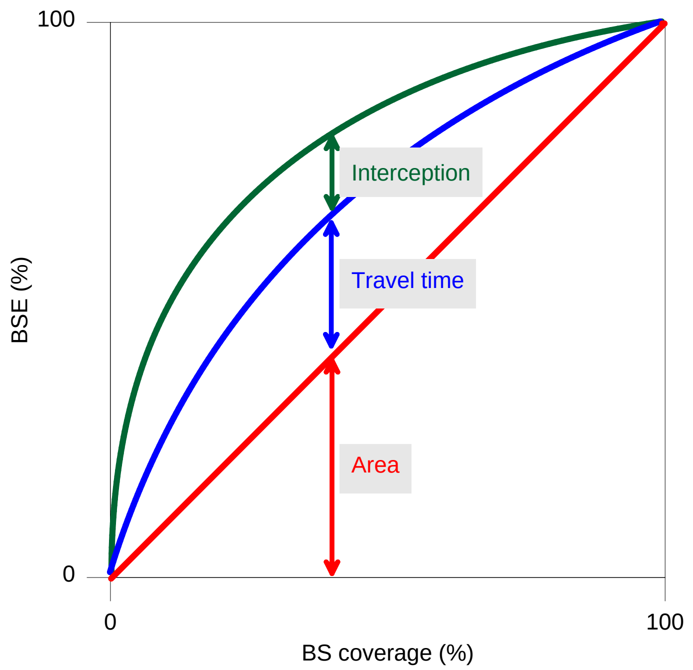


Figure 1.1
Theoretical effects of non-fertilized buffer strips, without background loads

At the beginning of the research project we formulated a hypothesis in which we distinguished three potential effects of non-fertilized buffer strips (Figure 1.1): 1) fertilization or area effect, 2) travel time effect, and 3) interception effect. This picture holds for a steady-state situation for a well-defined hydrological situation. In situations with a contribution of background loads (deposition, mineralization of soil organic matter, upward seepage from deeper aquifers, infiltration from surface water), maximum BSE will be less than 100%. For situations with downward seepage or upward seepage the hydrological discharge area is not only determined by the distance between ditches, and BSE will be less and more, respectively. In such cases the BS coverage is more difficult to define. For that reason we will later use absolute BS widths.

The fertilization effect is simply the effect of the area weighted lower fertilizer rate due to the presence of the non-fertilized strip. This effect is by definition proportional to the area fraction of the buffer strip (buffer strip width divided by distance from ditch to water divide), as indicated by the 1:1 line in Figure 1.1. There is no reason why this effect would not occur under Dutch circumstances, but it wouldn't make buffer strips more effective than simply reducing fertilizer rate on the whole field.

Both travel time and interception effect are specifically due to the positioning of the BS next to the ditch. The travel time of the discharging precipitation surplus from a strip next to the ditch is lower compared to the rest of the field. As retention of dissolved nutrients is normally positively correlated with time, higher concentrations may be expected in water discharging from a nearby strip, compared to water from the rest of the field further away from the ditch. Therefore, a reduction of fertilizer use in a nearby strip is expected to be more effective than further away from the ditch. The interception effect refers to retention of nutrients in lateral, mainly shallow runoff from the remainder of the field through the BS.

Our hypothesis is that BSE equals the fertilizer effect under prevailing Dutch circumstances, i.e. there is no specific buffer strip effect of travel time and interception. As data from field experiments will only yield the combined BS effect, we cannot distinguish between the three hypothesized sub-effects.

1.4 Goal

The main goal of this project was to collect experimental evidence on BSE in reducing nutrients loads to surface waters for five typical hydrogeological situations in the Netherlands. The second goal was to assess the expected range of BSE in the Netherlands with a model. The model should also be able to describe the long term evolution of BSE with time (i.e. not the seasonal variability due to weather). Thirdly, we investigated how cost-effectiveness of BS compares with alternative measures to reduce nutrient loads from agricultural land to surface waters.

1.5 Structure of the project and this report

The project was subdivided along the lines of the three project goals above, leading to three separate Chapters in this report: field study (Chapter 2), model study (Chapter 3) and cost-effectiveness (Chapter 4). The project parts were however strongly connected, according to Figure 1.2. A separate report on cost-effectiveness was released before (Noij et al., 2008). We discussed the over-all results of the project in the summary.

This report presents a concise description of the work done and the obtained results. For more detailed descriptions of the field research we refer to two scientific peer-reviewed papers published in a special issue on buffer strips of Journal of Environmental Quality and to a third submitted paper :

1. A novel method to determine buffer strip effectiveness on deep soils (Heinen et al., 2012; Journal of Environmental Quality 41(2): 334-347. doi:10.2134/jeq2010.0452).
2. Effectiveness of non-fertilized buffer strips to reduce nitrogen loads from agricultural lowland to surface waters (Noij et al., 2012; Journal of Environmental Quality 41(2): 322-333. doi:10.2134/jeq2010.0545).
3. Effectiveness of non-fertilized buffer strips to reduce phosphorus loads from agricultural lowland to surface waters (Noij et al., in prep.; submitted to Soil Use and Management).

The first paper focuses on the method to measure BSE that was specifically developed for deeply permeable soil. This method was used for all experimental locations. The second paper focuses on the results for N and the third paper on those for P.

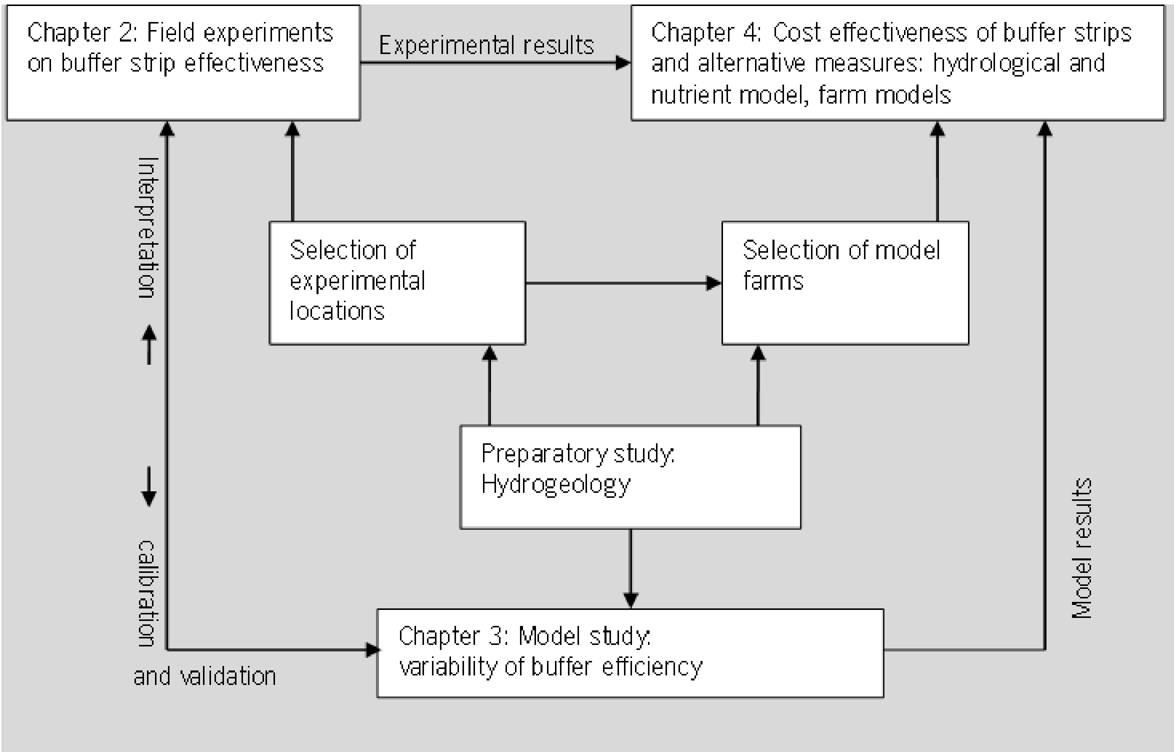


Figure 1.2
Over-all structure of the research project

Besides these scientific papers a special series of Alterra reports have appeared as results of this project, 'Effectiveness of buffer strips publication series', see page 5.

2 Field study

Box 2.1 Abbreviations and symbols used in Chapter 2

Symbol	Explanation	Units
^2H_2O	Deuterated water	
BS	Buffer strip	
BSE	Buffer strip effectiveness	dimensionless; %
BSE_I	BSE based on flow-averaged concentration	dimensionless; %
BSE_{II}	BSE based on average concentration	dimensionless; %
bss	Below soil surface	cm
C	Concentration	g m^{-3} (=mg L^{-1})
C_{gw}	Concentration in upper groundwater	g m^{-3}
$\overline{C}$	Flow-averaged concentration	g m^{-3}
Cl	Chloride	
DOC	Dissolved organic carbon	
Loc	Location (in statistical analysis)	
Load	Load	g
LS	Leaching season (in statistical analysis)	
MHG	Mean highest groundwater level (bss)	cm
MLG	Mean lowest groundwater level (bss)	cm
N	Nitrogen	
N_t	Total N	
N_{ts}	Total soluble N	
NH_4	Ammonium	
NO_3	Nitrate	
n	Number of samples	
n.d.	Not determined	
NHI	National Hydrological Model (Instrument)	
OM	Organic matter content	% (by weight)
P	Phosphorus	
P_t	Total P	
P_{ts}	Total soluble P	
PO_4	Ortho-phosphate	
PS	Precipitation surplus	mm
PSD	Phosphorus saturation degree	%
Q	Discharge	m^3 or mm
$Q_{0.5}$	Cumulative discharge after tracer application at $t_{0.5}$	m^3
Rep	Replicate (in statistical analysis)	
REF	Reference strip, reference treatment	
REML	Restricted (or residual) maximum likelihood	
TR	Tracer recovery	% (by weight)
$t_{0.5}$	Time after tracer application where half of final *TR* was reached	d
Treat	Treatment (in statistical analysis)	
wd	Water divide	m

2.1 Materials and methods

2.1.1 Selection of sites

Since we cannot do measurements at a large number of locations, we carried out a preparatory study to define characteristic hydrogeological conditions in the Netherlands. Van Bakel et al. (2007) divided the Netherlands into six hydrogeological classes, based on 1) ditch density, 2) depth and conductivity of the upper aquifer, 3) resistance of the aquitard below the upper aquifer, and 4) a compromise between more differentiation and lower number of classes (with larger area) (Figure 2.1; Table 2.1 and text below). In each of the classes a characteristic experimental location was selected.

a) Shallow sand: sandy aquifer with slope on impermeable subsoil <1 m below soil surface (bss). All precipitation surplus discharges through the shallow aquifer. Location: Winterswijk.
b) Deep sand: deep sandy aquifer (>4 m bss). Shallow discharge flow paths starting near the ditch, deeper flow paths starting further away from the ditch. Situations with downward seepage and/or regional flow are not depicted in Figure 2.1 but also pertain to this class. Location: Beltrum.
c) Sandy aquifer with less-permeable top soil. Ditch bottom in the aquifer. Predominant deeper flow paths. Not considered in this study because of the small area.
d) Holland peat: less-permeable deep aquifer in peat soil. Predominant shallow flow paths. Location: Zegveld.
e) Interrupted sand: deep sandy aquifer, interrupted by less permeable loam layer below the ditch bottom (1 - 4 m bss). Both shallow and deep flow: ratio depends on conductivity of aquifer and loam layer. Location: Loon op Zand.
f) Holland clay: less-permeable deep aquifer in clay soil. Tile drain discharge predominates because the majority of clay soils is tile drained. Location: Lelystad.

The six classes a) – f) cover 2.4, 33.5, 1.4, 12.8, 16.1 and 33.7% of total land area respectively (see also App. 8). Classes b), e), and a) provide a hydrogeological sequence with increasing proportion of shallow flow. Classes d) and f) belong to the so-called Holland profile. The sandy sites are drained by ditches (Beltrum, Loon op Zand) or a small modified natural stream (Winterswijk). Zegveld and Lelystad are situated below sea level in a polder with controlled water level. Tile drain spacing at Lelystad was 8 m. Zegveld is a moorland site with grazed grassland, drained by abundant parallel ditches (60 m apart). This area suffers from soil subsidence due to peat mineralization which has led to concave fields with elevated ditch borders. Therefore, these fields partly drain surface runoff through the middle and parallel to the ditch. Some basic soil information of the experimental sites is presented in Table 2.2.

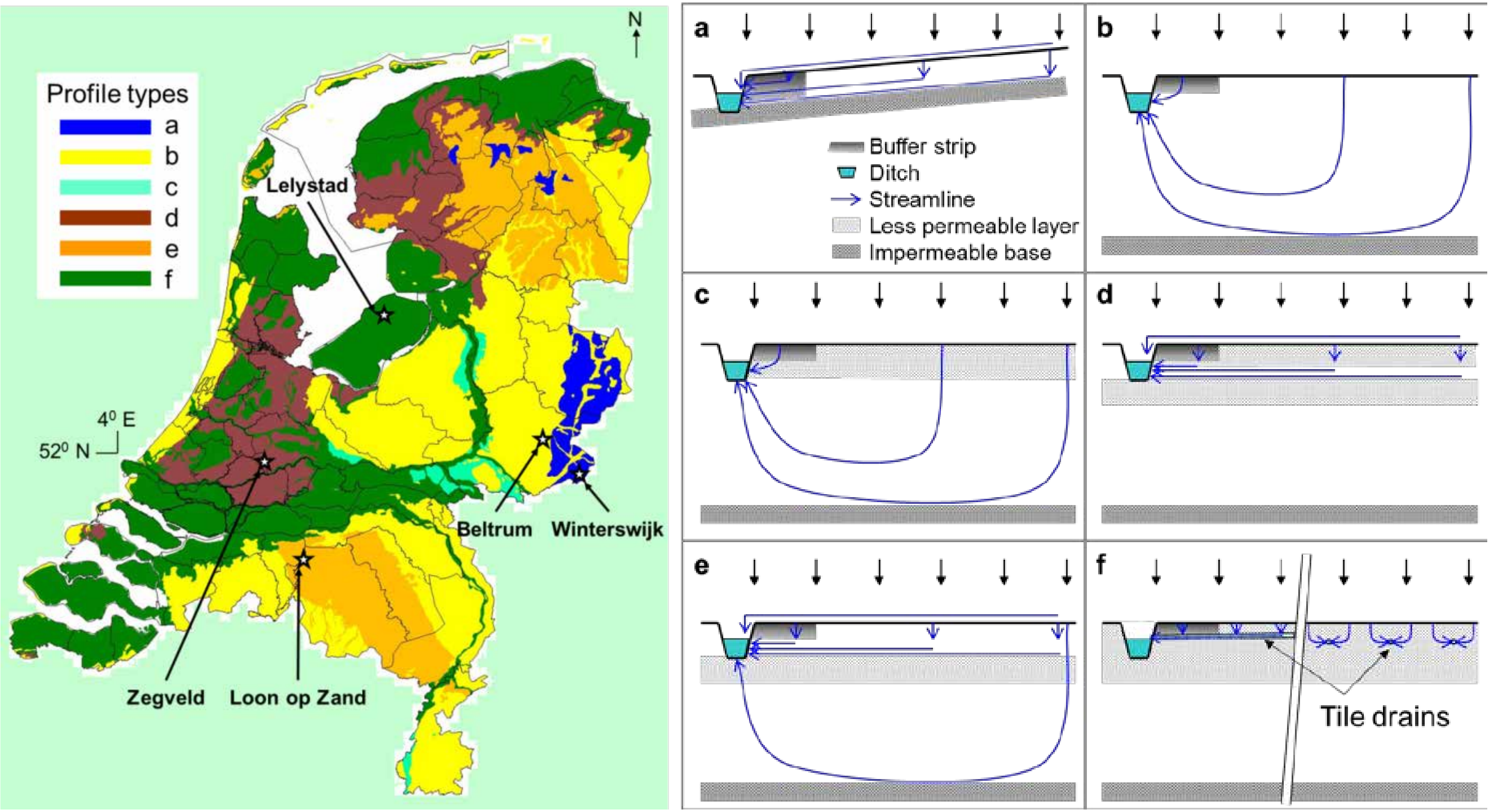


Figure 2.1

Geographical distribution (left) and profile (right) of hydrogeologic classes in the Netherlands (Van Bakel et al., 2007). Selected experimental locations (left) and expected flow paths (right). (a) Shallow sand: sandy aquifer with slope on impermeable subsoil <1 m below soil surface (bss), area 2.4%. (b) Deep sand: deep sandy aquifer, area 33.5%. (c) Sandy aquifer with less-permeable top soil, area 1.4%. (d) Holland peat: less-permeable deep aquifer in peat soil, area 12.8%. (e) Interrupted sand: deep sandy aquifer, interrupted by less-permeable loam layer, area 16.1%. (f) Holland clay: less-permeable deep aquifer in tile drained clay soil, area 33.7%. Area refers to total surface, for agricultural area see Appendix 8.

Table 2.1
Site characteristics

Site hydrogeology	Slope	Soil (FAO, 2002, 2008)	Land use	Water divide[a] m	MHG; MLG[b] cm bss	Ditch bottom[c] cm bss	Ditch water level in summer and winter[c] cm bss	Coordinates; elevation m above sea level
Beltrum **deep sand**	<1%	Sandy soil of periglacial aeolic origin (gleyic podzol)	Fodder maize; grass winter crop	60[d], 130[e]	40; 140	130	112; 126	52°04'56''N, 06°32'11''E; 17
Zegveld **Holland peat**	0	Peat soil (terric histosol)	Grassland	30[d,e]	25; 80	140	90; 90	52°8'22''N, 4°50'11''E; -3
Winterswijk **shallow sand**	2%	Sandy soil on boulder clay <1 m bss (Eutric Gleysol)	Grassland	80[f]	30; >200	138	125; 131	51°54'57''N, 6°43'22''E; 45
Loon op Zand **interrupted sand**	0	Sandy soil (Haplic Podzol)	Grassland	15[d], 75[e]	70; >180	154	132; 111	51°37'28''N, 5°5'36''E; 9
Lelystad **Holland clay**	0	Silty clay loam (Calcaric Fluvisol)	Maize	150[d]	70;120	137	128; 132	52°32'25''N, 5°33'021''E; -4

a The rainwater surplus at both sides of the water divide (wd) flows away in opposite directions. In sloping areas, wd is fixed by the highest contour line in the field, but in a plain it is the dynamic position of maximum elevation of the groundwater plane. It may be determined as the average maximum groundwater elevation in a transect perpendicular to the ditches, which is often located halfway between two ditches. The theoretical discharge area is calculated as wd × reservoir length (Figure 2.2).
b MHG, mean highest groundwater; MLG, mean lowest groundwater level in cm below soil surface (bss).
c As both levels are expressed in relation to soil surface level, the ditch water level measured from the bottom is calculated as ditch bottom – ditch water level.
d Top of measured groundwater level.
e Half the distance between two ditches.
f Top of the slope.

Table 2.2

Soil characteristics of the five experimental sites: organic matter content (OM), soil texture (on mineral basis), dry bulk density (r_d), pH, and phosphorus saturation degree (PSD). Also presented are the more labile properties mineral N content (N_{min}), Pw and P-AL as determined at the start of the experiment

Site	Replicate	Depth cm bss	OM % mass	<2 μm % mass	<16 μm % mass	<50 μm % mass	>50 μm % mass	ρ_d g cm^{-3}	pH-H_2O	PSD %	N_{min} mg kg^{-1}	Pw mg P_2O_5 l^{-1}	P-AL mg P_2O_5 hg^{-1}
Beltrum	A	0-30	5.7	1.9	4.0	7.3	92.7	1174	5.7	49.3	3.8	30.5	50.6
		30-100	3.4	2.2	3.1	6.5	93.5	1701	5.6	14.3	6.2	2.8	n.d.
	B	0-30	5.0	4.2	5.0	8.0	92.0	n.d.	6.0	52.7	1.9	32.0	57.1
		30-100	1.2	2.5	2.7	4.2	95.8	n.d.	5.9	8.4	1.9	1.4	n.d.
	C	0-30	5.5	3.3	5.7	9.1	90.9	n.d.	6.1	50.4	1.3	30.7	58.9
		30-100	1.6	4.2	4.9	7.0	93.0	n.d.	5.8	5.5	2.2	0.9	n.d.
Zegveld	A	0-30	52.0	70.8	83.2	83.3	16.7	562	5.0	11.1	47.8	11.3	15.4
		30-100	74.2	47.7	64.4	76.3	23.7	188	4.8	2.5	77.1	2.2	n.d.
	B	0-30	51.1	66.9	74.5	78.5	21.5	562	5.0	11.7	30.1	11.0	15.4
		30-100	75.8	64.0	73.5	72.5	27.5	188	5.0	3.4	66.8	2.3	n.d.
	C	0-30	48.8	63.8	76.8	75.5	24.5	562	5.3	9.9	25.4	6.4	9.3
		30-100	72.0	71.3	82.1	83.8	16.2	188	5.0	2.9	52.3	1.2	n.d.
Winterswijk	A	0-30	6.6	6.7	13.4	18.4	81.6	1428	6.1	41.8	7.4	41.5	55.9
		30-60	3.9	33.1	43.5	48.4	51.6	1408	5.6	17.4	3.1	12.9	n.d.
Loon op Zand	A	0-30	3.3	0.6	2.8	7.1	92.9	1549	5.9	60.9	2.3	43.4	39.2
		30-100	1.5	0.0	1.5	8.0	92.0	1568	5.5	34.9	2.0	11.3	n.d.
Lelystad	A	0-30	2.6	14.2	24.6	35.8	64.2	1586	7.2	17.9	2.8	39.8	38.0
		30-100	1.7	4.6	8.8	19.9	80.1	1386	8.2	6.1	3.3	6.6	n.d.

n.d.: not determined

2.1.2 Experimental approach and set-up

In order to measure BSE, we installed in-stream reservoirs in the ditch (Figures 2.2 and 2.3) to collect the discharging water from the adjacent soil, to determine its quantity and quality. The first outflow into the reservoirs contains historic water that is not yet influenced by the treatments. Therefore, we added a tracer at the outer edge of both treatments to determine its breakthrough curve and travel time. Only if the total measuring period exceeds this travel time, an effect of the BS can be expected and determined. Details follow below, and more details can be found in Heinen et al. (2012) and Noij et al. (2012).
At each location there were two treatments:

1) a 5-m-wide non-fertilized BS and,
2) a reference treatment (REF) treated like the rest of the field, including fertilization up to the ditch bank except for a small non-fertilized buffer strip of 25 cm (grassland) or 50 cm (arable land); according to current legislation [3].

We used reservoir observations to calculate accumulated loads and flow-weighted concentrations to assess BSE. We performed a statistical analysis of the results of individual samplings to assess BSE and its statistical significance.

To compare BSE based on reservoirs with BSE based on upper groundwater, the concentration of the upper groundwater was monitored underneath both treatments and in the field. The groundwater concentrations were also used for the sake of interpretation of reservoir results.

Between October 2005 and February 2006, paired treatments were installed at all locations: an non-fertilized grass BS and a reference strip (REF) cropped and managed like the adjacent field (Replicate A, Figure 2.2). Only at Beltrum and Zegveld, two extra paired treatments (Replicates B and C) were installed before the start of the second leaching season. At the grassland sites, the existing sward remained; at the maize sites, grass was sown in the BS to establish a normal sward. All grass strips (BS and grassland REF) were harvested and sampled for N withdrawal throughout the season, maize REF once a year. Like the rest of the field, the REF was cropped and managed according to farmers' practice, including slurry and fertilizer application, except for an obligatory uncultivated strip of 0.25 m (for grassland) or 0.5 m (for maize) from the edge of the ditch bank. Both treatments were installed along the ditch, 5 m wide and 15 m long. Opposite the centre of each treatment, a 5-m-long wooden reservoir, reaching to the middle of the ditch, collected all surface and subsurface discharge from the field. At Lelystad, the treatments were 25 m and the reservoirs 16 m long, so as to collect outflow from two subsurface drains. In Winterswijk, both reservoirs were enlarged in 2007 from 5 to 12.5 m long to increase the discharge area. Treatments were longer than reservoirs to prevent interaction between treatments (Figure 2.2). Reservoir walls consisted of 0.045 m thick tongue and groove planks driven down to approximately 1.5 m below the bottom of the ditch. Once a year, reservoirs were pumped empty for visual inspection of leakage through the walls. Except for Zegveld, we mounted additional walls of composite wood board with bentonite between the two walls to prevent any leakage. The water level in the reservoir was maintained at ditch water level by pumping out excess water (tolerance 0.01 m). In Zegveld and occasionally Loon op Zand, water had to be pumped in during summer to compensate for infiltration from ditch to soil. Figure 2.3 gives photographic impressions of the five experimental sites.

[3] LOTV: Lozingenbesluit Open Teelt en Veehouderij

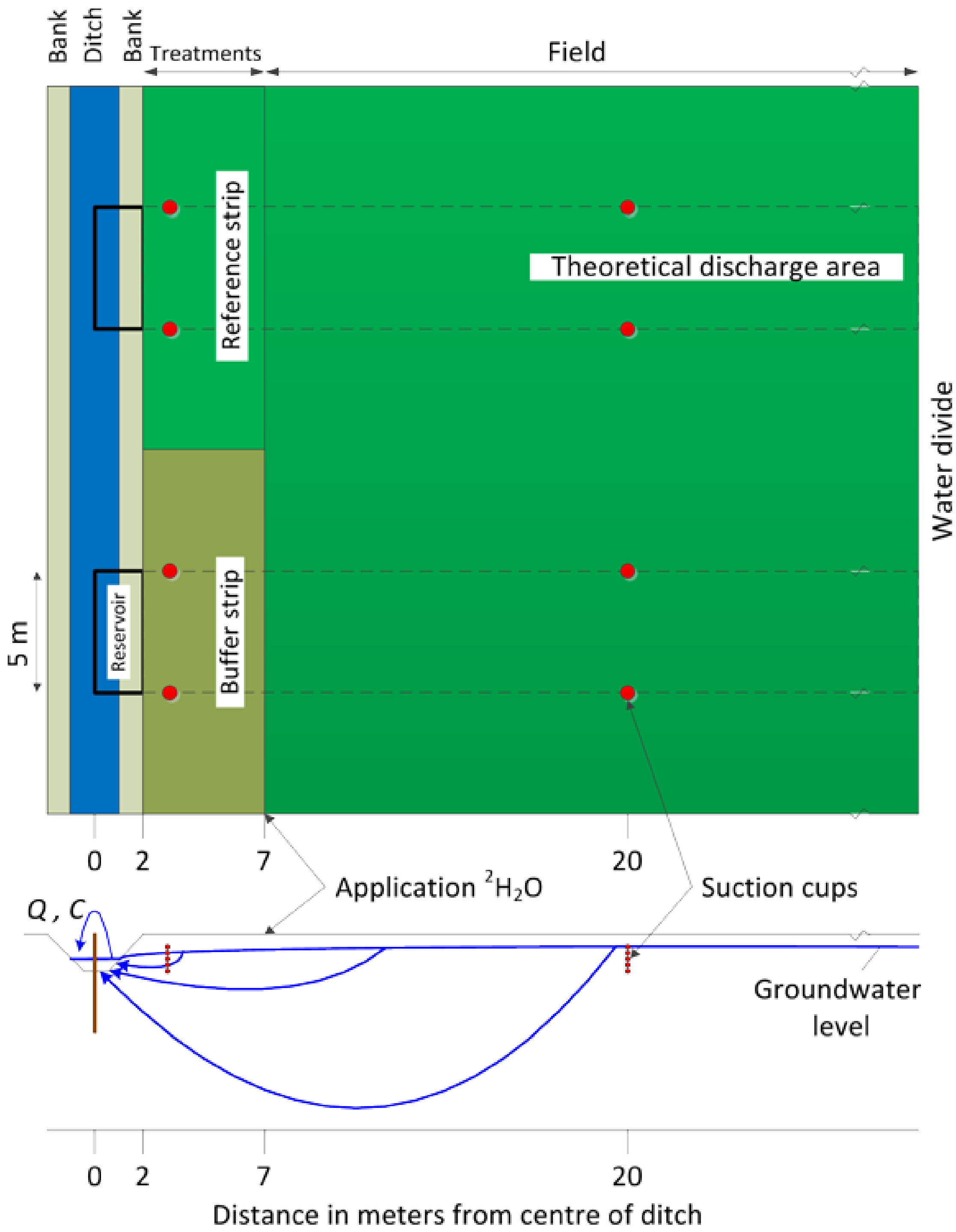


Figure 2.2
Aerial layout of one replicate of the experiments (above) and corresponding transect (below). At Beltrum and Zegveld additional suction cups at distances 4, 6 and 8 m. At Beltrum the last set of suction cups was at 50 m instead of 20 m. More details in Heinen et al. (2012) and Appendix 1

Figure 2.3
(Top) Photograph of the Beltrum experimental site. The ditch with reservoirs is at the right, the reference plot (REF) with maize planted up to the ditch is in the back, the 5-m-wide buffer strip (BS) is in the middle. The 3-m-wide grass strip in the front was for access to the experiment. (Below) Photographs of the four other experimental sites (before installation of the extra wood board walls) (clockwise): Winterswijk, Zegveld, Lelystad, Loon op Zand

2.1.3 Discharge and tracers

Discharge (Q, m^3) from the reservoir was measured at the pump outlet with a flow meter and logged by a programmable data taker that activated an automatic sampler at fixed discharge amounts to take water samples from the reservoir (see also next subsection).

Measured Q in the reservoirs was divided by discharge area to obtain Q in millimeters. Discharge area was calculated as the reservoir length (5 m, 12.5, or 16 m) times the distance to the water divide (Table 2.1) based on the position of the maximum measured groundwater level, except in Winterswijk, where it was based on the top of the slope (Table 2.1). Discharge Q (mm) was further divided by precipitation surplus (PS) to calculate the ratio Q/PS. Precipitation was measured on site, and estimated reference (Makkink) evapotranspiration was taken from the nearest weather station of the Royal Netherlands Meteorological Institute (KNMI). If $Q/PS < 1$, part of PS is lost from the observed system, whereas $Q/PS > 1$ indicates input from other sources than PS.

To assess the hydrological lag time of the treatment response, we applied a conservative tracer (deuterated water, ^2H_2O or D_2O) at the edge of the treatment strips (6.5 m from ditch centre; Figure 2.2) before the first leaching season. The D_2O concentration of the discharge water from the reservoirs was measured and compared with the measured local background concentration to detect breakthrough. The cumulative breakthrough (load) was plotted as a function of time or cumulative discharge to determine the mean time needed for 50% recovery. Further details can be found in Heinen et al. (2012).

2.1.4 Reservoir concentrations and loads

Water samples were taken proportional to discharge which is required for establishing loads or flow-averaged concentrations (e.g., De Vos, 2001; Rozemeijer et al., 2010a). Sampling bottles were filled in five steps, each step corresponding to approximately 1 mm of precipitation surplus. Water samples were immediately stored in an on-site refrigerator (<4 °C) and transported to the laboratory once a week for analysis (including partly filled bottles); if no water sample was taken automatically, a sample from the reservoir was taken manually (if water was present).

Reservoir water samples were split into three subsamples after thorough mixing. The first unfiltered subsample was analysed for total nitrogen (N_t) and total phosphorus (P_t) with a segmented flow analyser (SFA) after persulfate-borate destruction (NEMI I-4650-03 and I-2650-03; www.nemi.gov). The second subsample was analysed in the same way, but after filtering over 0.45 mm (Whatman RC55 regenerated cellulosis membrane) to measure total soluble N (N_{ts}) and total soluble P (P_{ts}). The third subsample was filtered likewise and analysed for NO_3–N[4], NH_4–N, PO_4–P (all in 0.01 M $CaCl_2$ with segmented flow analyser, SFA), dissolved organic carbon (DOC) (SFA) and Cl (flow injection analyser, FIA). Organic nitrogen concentrations N_{org} were calculated as $N_{org} = N_t – NO_3 – NH_4$. Here, we focus on N_t, P_t, NO_3–N, PO_4–P and Cl only. Results for N_{ts}, P_{ts}, NH_4–N, N_{org}, and DOC are given in Appendix 3 without further discussion.

Loads (g) from soil to reservoirs were computed as the product of Q and concentration C (g m^{-3}):

$$Load = \sum QC \qquad [2.1]$$

[4] Including NO_2.N

The flow-averaged leaching concentration, $\overline{C}$ (g m^{-3}), was computed according to (e.g., Chaubey et al., 1994; 1995; Heinen et al., 2012):

$$\overline{C} = \frac{Load}{\sum Q} \qquad [2.2]$$

In Eqs. [2.1] and [2.2], the summation sign refers either to a period of equal time or to a period of equal discharge. In this study sums were calculated for periods of equal discharge, instead of more common periods of equal time, to reduce the influence of spatial variation in discharge between the treatments (Heinen et al., 2012). For each leaching season and each pair of treatments, the lowest discharge at the end of the leaching period was used. We also computed an average $\overline{C}$ for all leaching seasons (c.f. Heinen et al., 2012). Dutch leaching seasons typically run from 1 October until 1 April, but actual start and end dates were used.

2.1.5 Groundwater measurements

Polyester acrylate suction cups (porosity 65%, pore diameter 0.45 mm, inert to N and P) were installed in the soil inside both treatments and in the adjacent field (Figure 2.2). Cups were placed at five depths covering the range between mean highest and mean lowest groundwater levels as presented in Table 2.1. At Beltrum and Zegveld the cups were positioned at five distances from the centre of the ditch: 2, 4, 6, 8 and 20 (Zegveld) or 50 m (Beltrum). At the other three locations the positions were at 2 and 20 m only. The sampling frequency was six to seven times per leaching season and two to three times during the summer season. In Replicate A of Beltrum, sampling ran from December 2006 to the end of the experiment (total 32). For Beltrum B and C, sampling started November 2007 (total 22) and for all other locations in 2008 (Zegveld 13, Loon op Zand 18, Winterswijk 6, and Lelystad 10). The first cup just below the groundwater level was used to sample upper groundwater, except in Winterswijk, where groundwater levels were too dynamic. Here we used the cups of 80 and 100 cm bss. Water samples from the suction cups were analysed for N_{ts}, NO_3–N[4], NH_4–N and PO_4–P, DOC, and Cl. Groundwater concentrations are denoted as C_{gw}.

2.1.6 Analysis of Buffer Strip Effectiveness (BSE)

Heinen et al. (2012; main text and their Appendices) presented a thorough analysis of how to determine buffer strip effectiveness (BSE). For the current study BSE based on reservoirs was computed in two different ways. The first calculation, BSE_I, was based on $\overline{C}$ according to:

$$\mathrm{BSE_I} = 1 - \frac{\overline{C}_{BS}}{\overline{C}_{REF}} \qquad [2.3]$$

The second calculation, BSE_{II}, was based on average concentrations C_{avg} of the separate reservoir concentration measurements, resulting from the statistical analysis described below:

$$\mathrm{BSE_{II}} = 1 - \frac{C_{\mathrm{BS,avg}}}{C_{\mathrm{REF,avg}}} \qquad [2.4]$$

For statistical analysis a restricted (or residual) maximum likelihood analysis (VSNI, 2010; directive REML in GenStat) was conducted by J.T.N.M. Thissen of Biometris (Wageningen UR, Plant Research International). The fixed model in REML was defined as:

constant + Loc + Treat + LS + Loc×Treat + Loc×LS + Treat×LS + Loc×Treat×LS, and the random model was:

$$Loc \times Rep + Loc \times Rep \times Treat + Loc \times Rep \times LS + Loc \times Rep \times Treat \times LS,$$

where Loc(ation) = Beltrum, Zegveld, Winterswijk, Loon op Zand and Lelystad, Treat(ment) = BS and REF; Rep(licate) = A, B and C, and the leaching seasons LS were 2006 - 2007, 2007 - 2008, 2008 - 2009 and 2009 - 2010. We tested the null hypothesis that there is no difference between REF and BS. The *F* probability (*P*) values obtained for the fixed model terms were used to assess significance: $P < 0.05$. Inspection of the distribution of the residuals of P_t *(Load* and *C)* revealed that the residuals were not normally distributed. Therefore, the REML analysis for P_t was repeated with log-transformed data for which better residuals were obtained. Both an integrated statistical analysis for all locations and replicates and partial analyses were performed for the two sites with three replicates (Beltrum, Zegveld). In the partial analyses all terms with Loc were, of course, removed from the model.

2.2 Results

2.2.1 Discharge and tracer breakthrough

During the leaching seasons (Oct - Apr) at all locations there was rainfall excess at the soil surface (Appendix 1), indicating that leaching into the reservoirs should occur. The total discharge during the leaching seasons and the corresponding precipitation surplus is listed in Table 2.3. At Beltrum the ration *Q*/*PS* was lower than one. Apparently the recharge area was less than anticipated. This was confirmed by Heinen et al. (2012) with the help of a flow path analysis model. Only rainfall excess from the first approximate 30 m next to the ditch discharges into the ditch.

During the study it became quickly evident that the recharge area at Loon op Zand was very small, on the order of 15 m. This may have affected the deuterium breakthrough, and was also the reason for the additional study on the representativeness of this location for the hydrogeology class e) (Hoogland et al., 2010).

At Winterswijk discharge was very fast and different between BS and REF from the beginning. Ground-penetrating radar measurements revealed erratic gullies in the top of the impermeable boulder clay layer (Heinen and Van Kekem, 2011). These may have been the cause of the observed difference in discharge between the two treatments. Therefore we decided to enlarge the treatments and reservoirs (from 5 to 12.5 m) in 2007 to reduce this effect. But, the opposite occurred. Between 2007 and 2010, discharge at BS was twice as much compared with REF during the leaching seasons. The apparent discharge area of the BS must have been much larger compared with REF. The top of the slope was about 80 m from the stream, but the contour lines deviated away from the stream opposite the BS, which likely caused extra discharge to the reservoir of the BS.

At Zegveld and Lelystad the average *Q*/*PS* ratio indicated almost complete recovery of *PS* in the reservoirs.

Surface runoff was not observed at Beltrum, and could not occur at Lelystad and Loon op Zand because of the presence of an elevated border at the start of the ditch bank. At Winterswijk only a few occasions of surface runoff were observed (Appels et al., 2011), and at Zegveld every now and then ponding occurred.

Table 2.3

Final tracer recovery (TR), number of days ($t_{0.5}$) and accumulated amount of water discharge ($Q_{0.5}$) since tracer application at half of final TR. Sum of discharge during leaching seasons for which BSE was calculated (Q), corresponding precipitation surplus (PS) and number of flow weighted reservoir samples (n). Q in mm and PS are based on water divides as given in Table 2.1

Site Hydrogeology	Replicate	Period	Treatment	*TR* %	$t_{0.5}$ d	$Q_{0.5}$ m^3	*Q* m^3	*Q* mm	*PS* mm	*Q/PS*	 avg	*n*
Beltrum	A	2006-2010	REF	40.0	146	45	132	439	866	0.51	0.61	153
Deep sand			BS	36.3	411	51	133	443	866	0.51		152
	A	2007-2010	REF				82	272	538	0.51	0.61	97
			BS				88	294	538	0.55		93
	B	2007-2010	REF	66.9	136	28	90	299	542	0.55		109
			BS	36.9	790	63	95	317	542	0.58		84
	C	2007-2010	REF	97.8	496	65	119	395	502	0.79		92
			BS	63.3	778	71	107	357	502	0.71		96
Zegveld	A	2006-2010	REF	32.1	147	67	164	1097	1106	0.99	0.89	340
Holland peat			BS	32.2	260	87	180	1200	1106	1.09		333
	A	2007-2010	REF				80	636	744	0.72	0.83	197
			BS				110	732	744	0.98		182
	B	2007-2010	REF	2.1	771	102	110	734	752	0.98		199
			BS	6.8	586	83	115	765	752	1.02		206
	C	2007-2010	REF	1.1	677	58	60	403	758	0.53		149
			BS	5.3	771	85	86	574	758	0.76		183
Winterswijk	A	2006-2007	REF	28.7	140	78	88	220	471	0.47	0.55	66
Shallow sand			BS	78.4	84	43	120	300	471	0.64		86
		2007-2010	REF	45.1			410	410	860	0.48	0.72	147
		Enlarged reservoirs	BS	89.0			820	820	860	0.95		268
Loon op Zand	A	2006-2009	REF	1.3	663	39	38	512	752	0.68	0.66	156
Interrupted sand			BS	1.6	467	26	36	486	752	0.65		156
Lelystad	A	2006-2009	REF	21.1	692	1617	1958	816	779	1.05	1.11	217
Holland clay			BS	20.8	114	820	2191	913	779	1.17		308

Figure 2.4 presents an example of the ^2H_2O breakthrough curves obtained for the Beltrum site (Appendix 2 gives those for the other four sites). There is clear spatial hydrological variability along the ditch as the curves are not identical for the treatments and replicates. This was observed for all locations. Incomplete recovery of ^2H_2O was expected because the deuterated water was subject to removal with time due to uptake by the roots (transpiration) and evaporation from the soil (Braud et al., 2005). Since we were only interested in the time of breakthrough, it is not necessary to quantify these losses. Assuming that the point of maximum recovery was reached at the end of the experiment, the average travel time of the water leaching from the edge of the treatment strip to the ditch was derived from the breakthrough curves as the time period after which half of the maximum recovery was observed (Table 2.3). After one or two leaching seasons breakthrough was observed at all locations. This indicates that measuring for three to four leaching seasons should have been enough to establish treatment effects in the discharge at all locations.

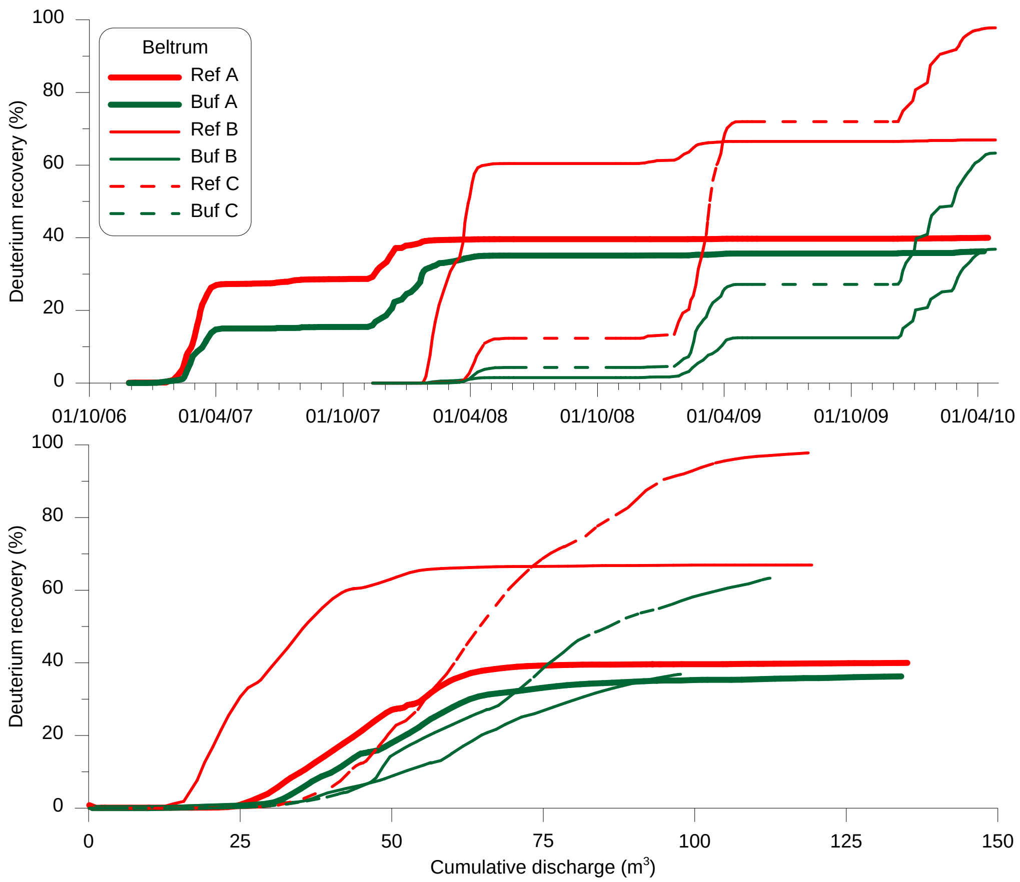


Figure 2.4

Example of the deuterium breakthrough curves obtained at Beltrum as a function of time (top) or cumulative discharge (bottom)

The breakthrough curves obtained for the other locations are presented in Appendix 2.

2.2.2 Nitrogen

For all locations the REF treatments had a (much) higher N surplus than the BS treatments (Table 2.4), indicating that at the soil surface a clear treatment effect was established. In what follows we first discuss the findings in the groundwater and end with the observations in the reservoirs.

Table 2.4

Average nitrogen and phosphorus balance (kg h^{1} y^{1}) for the buffer (BS) and reference (REF) treatments at the five locations

Locations	Crop	BS		REF					
		N	P	N			P		
		surplus	surplus	Fertilizer Rate	Crop Removal	Surplus	Fetilizer rate	Crop Removal	Surplus
Beltrum	Maize	**-65**	**-11**	174	139	**36**	35	20	**16**
Zegveld	Grass	**-280**	**-30**	288	368	**-80**	19	37	**-19**
Winterswijk	Grass	**-217**	**-39**	425	374	**51**	32	54	**-21**
Loon op Zand	Grass	**-134**	**-24**	316	325	**-4**	33	43	**-7**
Lelystad	Maize	**-29**	**-6**	171	168	**3**	45	31	**14**

Except for Loon op Zand, there are no differences in C_{gw} NO_3-N in the field (Table 2.5). However, except for Zegveld, we do see a decrease in C_{gw} NO_3-N underneath the BS with respect to REF. For Beltrum where groundwater was sampled at five distances from the centre of the ditch it can be nicely seen that this change abruptly occurred at the boundary of the BS (Figure 2.5). The concentrations in the reservoirs at the same sampling moments were not different (Figure 2.5). Especially for REF, the concentration in the reservoir does not resemble the concentration in the upper groundwater next to the ditch. A more thorough analysis on the Beltrum ground water samplings can be found in Heinen et al*.* (2012). Some additional information on upper groundwater data can be found in Appendix 4.

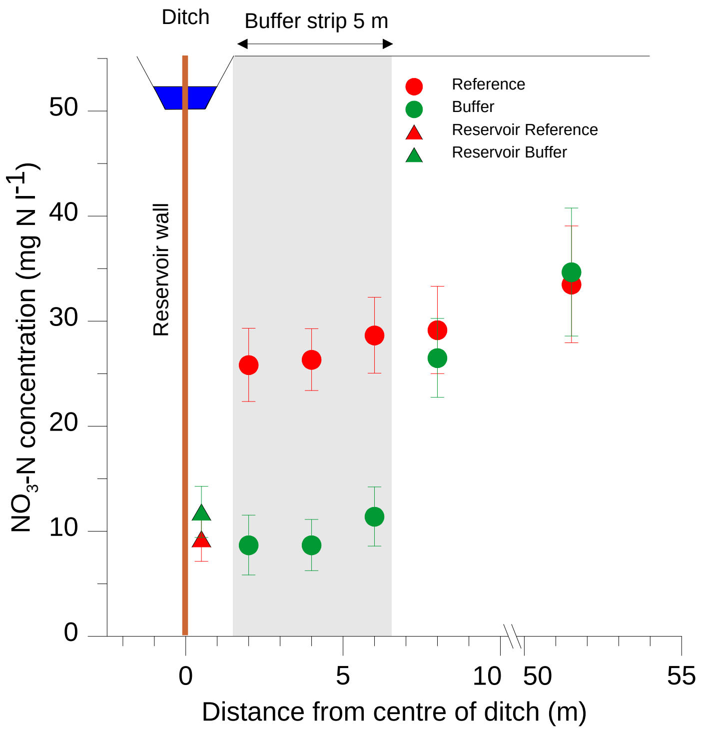


Figure 2.5

Time-averaged C_{gw} NO_3–N (mostly at 1 m bss; circles) as a function of distance from the centre of the ditch (shaded area represents the BS) for all treatments and replicates in Beltrum. Results for 0.25 m (triangles) represent average concentrations in the reservoirs at moments corresponding with groundwater sampling. The length of the error bars represents twice the standard error of the mean based on 32 (replicate A) or 22 (replicates B and C) samplings

Figure 2.6 presents the average N_t concentrations in the reservoirs for the whole experimental period summarized as Box-Whisker plots. The dynamic time courses can be found in Appendix 3. Table 2.5 lists $\overline{C}$ and C_{gw} at two distances from the centre of the ditch for NO_3-N and Cl and their ratio. As Cl is assumed to be inert, a lower ratio for the BS treatment can be regarded as an indication for extra NO_3-N removal (denitrification, uptake) underneath the BS. Comparing $\overline{C}$ and C_{gw} in Table 2.5 reveals that the concentrations in the reservoirs are not equal to that in the upper groundwater.

Table 2.5

Flow-weighted average concentration in both reservoirs ($\overline{C}$) and average upper groundwater concentration (C_{gw}) below treatments BS and REF and adjacent field opposite BS and REF (Figure 2.2). Standard deviations (±) for reservoirs in Beltrum and Zegveld are based on replicates, for upper groundwater on separate samplings

Location	**Period**	**Species**	**$\overline{C}$**		**C_{gw}, treatments**		**C_{gw}, field**	
			Ref	**BS**	**Ref**	**BS**	**Ref**	**BS**
Beltrum	2006/2010	NO_3-N	14.02 ± 4.72	16.43 ± 2.75	25.84 ± 12.69	8.69 ± 10.36	29.16 ± 15.27	26.50 ± 13.80
		Cl	25.70 ± 6.44	22.75 ± 3.63	16.86 ± 7.21	6.75 ± 4.71	18.27 ± 11.45	17.50 ± 11.46
		NO_3-N:Cl	0.55 ± 0.23	0.72 ± 0.17	1.53 ± 1.00	1.29 ± 1.78	1.60 ± 1.30	1.51 ± 1.27
Zegveld	2008/2010	N_{ts}[1]	7.99 ± 5.07	7.44 ± 4.81	11.78 ± 13.76	13.21 ± 9.75	11.33 ± 6.38	8.78 ± 7.61
		NO_3-N	1.12 ± 1.12	0.63 ± 0.72	7.66 ± 13.23	8.64 ± 9.70	5.77 ± 5.79	3.39 ± 7.03
		Cl	34.66 ± 21.54	34.21 ± 21.04	17.71 ± 4.02	16.77 ± 6.02	22.67 ± 7.81	19.96 ± 7.06
		NO_3-N:Cl	0.03 ± 0.04	0.02 ± 0.02	0.43 ± 0.75	0.52 ± 0.61	0.25 ± 0.27	0.17 ± 0.36
Winterswijk	2008/2010	NO_3-N	3.33	7.12	4.52 ± 2.66	2.18 ± 3.61	17.14 ± 13.35	14.14 ± 5.48
		Cl	20.81	19.59	18.04 ± 3.23	15.54 ± 5.38	23.92 ± 11.06	15.54 ± 5.34
		NO_3-N:Cl	0.16	0.36	0.25 ± 0.15	0.14 ± 0.24	0.72 ± 0.65	0.91 ± 0.47
Loon op Zand	2008/2009	NO_3-N	3.99	2.97	5.11 ± 4.78	0.40 ± 0.35	19.97 ± 8.94	4.05 ± 1.15
		Cl	22.06	19.86	14.97 ± 8.73	6.04 ± 1.12	17.12 ± 9.00	17.54 ± 8.55
		NO_3-N:Cl	0.18	0.15	0.34 ± 0.38	0.07 ± 0.06	1.17 ± 0.81	0.23 ± 0.13
Lelystad	2008/2009	NO_3-N	2.41	2.06	0.91 ± 1.10	0.23 ± 0.57	5.83 ± 5.04	2.30 ± 2.66
		Cl	102.49	94.31	13.18 ± 3.60	17.24 ± 14.79	14.30 ± 3.22	16.27 ± 15.23
		NO_3-N:Cl	0.02	0.02	0.07 ± 0.09	0.01 ± 0.04	0.41 ± 0.36	0.14 ± 0.21

[1] For Zegveld N_{ts} is also given as NO_3-N is not the dominant N species at this location

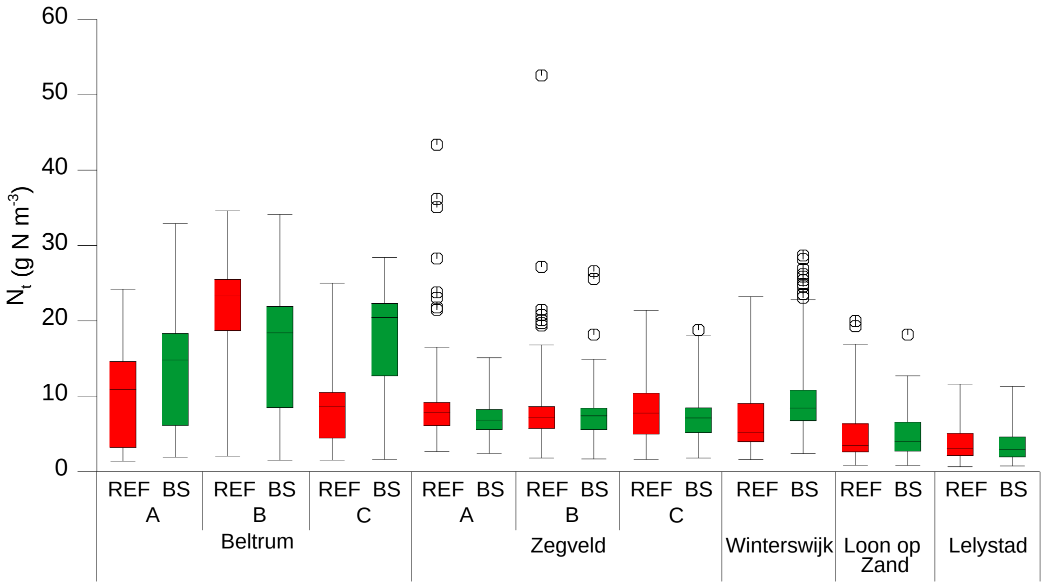


Figure 2.6

Box-whisker plots of the N_t concentrations in the reservoirs at all locations: minimum, first, second (median) and third quartile, and maximum values. Outliers (symbols) are shown which lie more than three times the box length from the box edge

The BSE_I for N_t is shown in Figure 2.7, with the corresponding difference in N_t between the treatments. For the total experimental period the BSE_I is negative for Beltrum and Winterswijk, and positive but low for Zegveld, Lelystad and Loon op Zand. The difference between BSE_I and BSE_{II} is small (Table 2.6). There is no statistically significant treatment effect in the overall analysis (Table 2.7). However, there is a statistically significant treatment effect for N_t (P = 0.005; BSE_{II} = 15.1%) in the separate analysis for Zegveld. This is not the case for NO_3-N and Cl (Table 2.7).

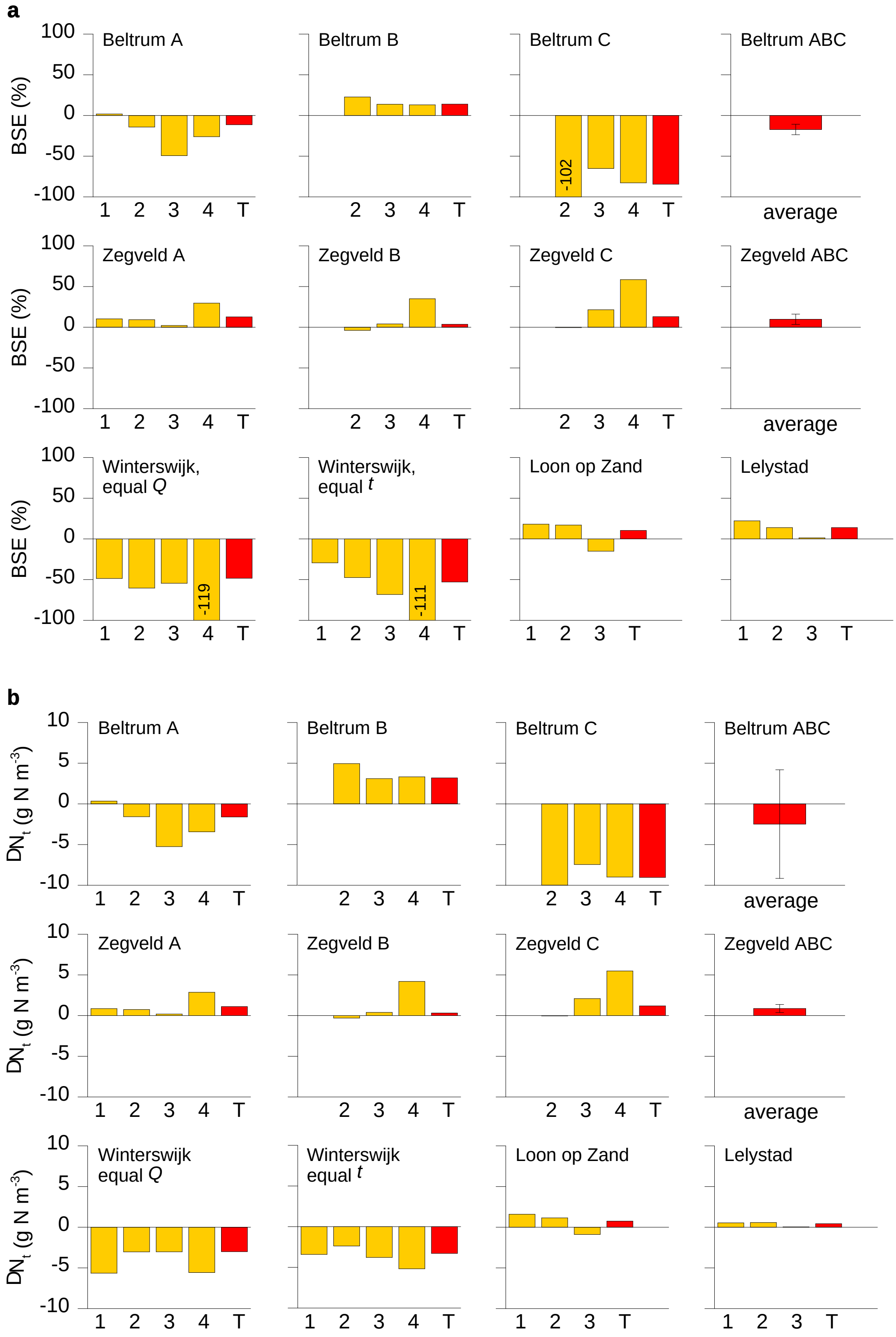


Figure 2.7

a) *Buffer strip effectiveness (BSE_i, Eq. [2.3]) for flow-weighted average N_t and* ***b)*** *absolute difference in N_t between REF and BS (DN_t), both for individual leaching seasons (1,2,3,4) and for the total experimental period (T), for all locations and replicates, and for periods of equal discharge and for Winterswijk also for periods of equal time*

Table 2.6

Buffer strip effectiveness (%) for five locations, based on flow averaged reservoir concentration (BSE_I; Eq. [2.3]) and based on statistical analysis of individual reservoir concentration measurements (BSE_{II}; Eq. [2.4]). For Beltrum and Zegveld the standard deviations (±) for BSE_I refer to the three replicates

Species	BSE	Beltrum	Zegveld	Winterswijk	Loon op Zand	Lelystad
N_t	I	-17.2 ± 6.4	9.8 ± 6.3	-48.3	10.4	13.9
	II	-14.8	15.1	-61.6	5.9	5.1
N_{ts}	I	-16.6 ± 6.2	6.9 ± 4.4	-62.5	9.9	13.2
	II	n.d.	n.d.	n.d.	n.d.	n.d.
NO_3-N	I	-17.2 ± 6.4	44.1 ± 43.8	-113.6	25.6	14.5
	II	-14.8	42.3	-147.1	23.9	8.0
Cl	I	11.5 ± 3.4	1.3 ± 0.8	5.9	10.0	8.0
	II	9.1	1.6	4.3	7.4	10.8

n.d.: not determined

Table 2.7

Buffer strip effectiveness, based on statistical analysis of individual reservoir concentration measurements (BSE_{II}; Eq. [2.4]), and corresponding average difference in reservoir concentration (C_{avg}) between both treatments (REF-BS). P-values for the terms loc(ation), treat(ment), leaching season (LS) and their mutual interactions. For the shaded cells P < 0.05

Analysis for	Species	REF–BS	BSE_{II}	P values						
		C_{avg} g m^{-3}	%	Loc	Treat	LS	Loc×Treat	Loc×LS	Treat×LS	Loc×Treat×LS
All locations	N_t	-0.68	-6.8	0.061	0.644	0.008	0.848	0.022	0.592	0.392
	NO_3–N	-0.94	-15.3	0.025	0.525	0.001	0.846	0.005	0.479	0.772
	Cl	2.58	6.8	<0.001	0.046	<0.001	0.118	<0.001	0.235	0.005
Winterswijk	N_t	-3.79	-61.6							
Loon op Zand	N_t	0.40	5.9							
Lelystad	N_t	0.16	5.1							
Only Zegveld	N_t	1.36	15.1		0.005	0.841			0.044	
	NO_3–N	0.44	42.3		0.257	0.169			0.179	
	Cl	0.59	1.6		0.790	0.001			0.342	
Only Beltrum	N_t	-2.34	-14.8		0.583	0.023			0.649	
	NO_3–N	-2.13	-14.8		0.578	0.001			0.573	
	Cl	2.34	9.1		0.262	0.007			0.649	

2.2.3 Phosphorus

For all locations the REF treatments had a higher P surplus than the BS treatments (Table 2.4), indicating that at the soil surface a clear treatment effect was established. In what follows we first discuss the findings in the groundwater and end with the observations in the reservoirs.

We could not detect a treatment effect on upper groundwater PO_4-P concentrations (C_{gw}), because C_{gw} was too low. Only 24 out of 222 samples (and only three out of 111 paired samples REF and BS) were above the detection limit (0.02 g m^{-3} PO_4-P; Appendix 5). Therefore, no P data are presented here for the groundwater samplings. Fraters et al. (2008) also found such low values below 148 sandy soil farms in the Netherlands (median C_{gw} < 0.06 g m^{-3} PO_4-P, which was their detection limit).

Figure 2.8 presents the average P_t concentrations in the reservoirs for the whole experimental period summarized as Box-Whisker plots. The dynamic time courses can be found in Appendix 4.

Median P_t concentrations ranged from 0.03 g m^{-3} in Lelystad to 0.42 g m^{-3} for REF in Winterswijk. Reservoir concentrations were temporally quite variable with outliers reaching 3 g m^{-3} P_t. We observed no clear periodicity. Median values for Beltrum and Lelystad compare well with those for ditch water of eleven farms on sandy soil in the Netherlands (<0.06 g m^3 P_t; Fraters et al., 2008). However, P_t and P_{ts} reservoir concentrations were clearly higher at the other sites on sandy soils (Loon op Zand and Winterswijk), and for the peat site Zegveld (Table 2.8).

Flow weighted average P_t, P_{ts} and PO_4-P concentrations in the reservoirs and BSE_l are given in Table 2.8. Figure 2.9 shows BSE_l and the difference in P_t between the treatments for the individual leaching seasons and for the total experimental period. For the total period all locations yielded a positive value for BSE_l.

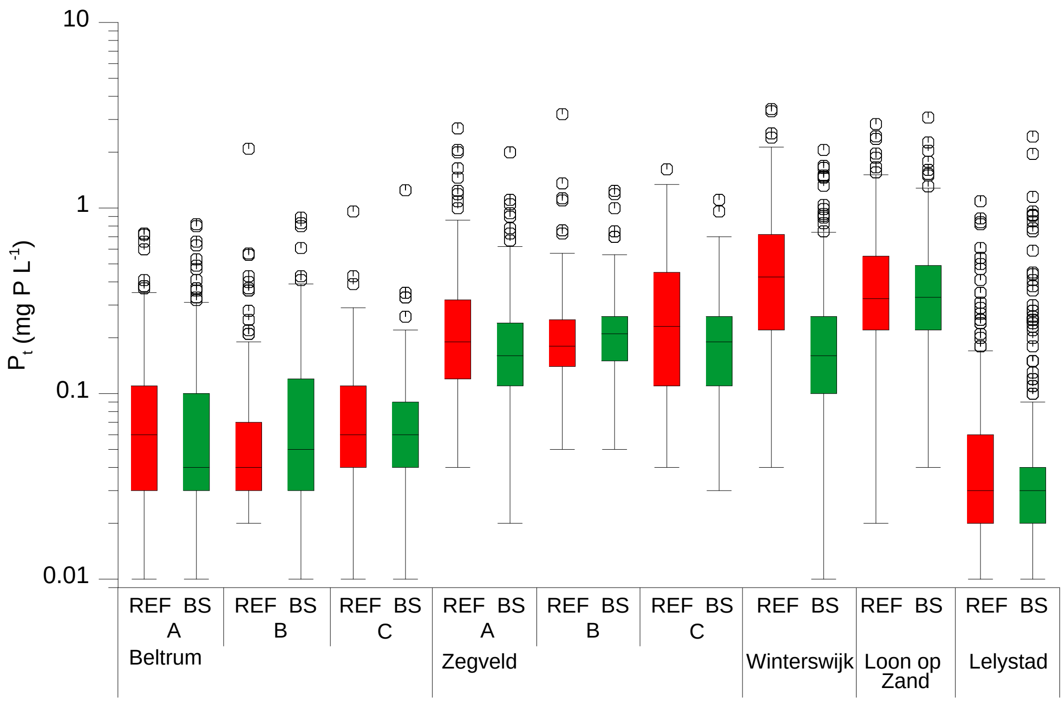


Figure 2.8

Box-whisker plots of the P_t concentrations in the reservoirs at all locations: minimum, first, second (median) and third quartile, and maximum values. Outliers (symbols) are shown which lie more than three times the box length from the box edge. Note the log-scale

Table 2.8

Flow weighted average P concentrations in REF and BS reservoirs (g m^{3}) and buffer strip effectiveness (BSE_l, %; Eq. [2.3]). For Beltrum and Zegveld standard deviations (±) for reservoirs are based on the three replicates

Species	REF, BS, BSE_l	Beltrum	Zegveld	Winterswijk	Loon op Zand	Lelystad
P_t	REF	0.078 ± 0.012	0.251 ± 0.138	0.536	0.527	0.034
	BS	0.066 ± 0.017	0.219 ± 0.132	0.208	0.504	0.028
	BSE_l	15.45 ± 4.64	12.73 ± 7.01	61.17	16.54	4.36
P_{ts}	REF	0.019 ± 0.006	0.146 ± 0.071	0.114	0.234	0.024
	BS	0.019 ± 0.005	0.134 ± 0.083	0.052	0.196	0.019
	BSE_l	3.64 ± 1.50	8.44 ± 4.11	54.58	20.76	16.50
PO_4-P	REF	0.006 ± 0.008	0.093 ± 0.044	0.056	0.156	0.011
	BS	0.001 ± 0.000	0.082 ± 0.052	0.021	0.112	0.008
	BSE_l	87.57 ± 124.4	11.50 ± 5.47	62.67	31.25	28.37

Table 2.9

Buffer strip effectiveness for Pt, based on statistical analysis of individual reservoir concentration measurements (BSE_{II}; Eq. [2.4]), and corresponding average difference in reservoir concentration (C_{avg}) between both treatments (REF-BS). P-values for the terms Loc(ation), treat(ment), leaching season (LS), and their mutual interactions. For the shaded cells P < 0.05

Analysis for	REF–BS C_{avg} g m^{-3}	BSE_{II} %	P-values						
			Loc	Treat	LS	Loc×Treat	Loc×LS	Treat×LS	Loc×Treat×LS
All locations	0.0227	17.7	<0.001	0.010	0.013	0.031	0.293	0.997	0.289
Winterswijk	0.2312	56.9							
Loon op Zand	0.0013	0.3							
Lelystad	-0.0012	-4.3							
Only Zegveld	0.0386	16.9		0.207	0.487			0.138	
Only Beltrum	0.0059	11.8		0.319	0.063			0.745	

At Beltrum we found a negligible absolute treatment effect of 0.012 g m^{-3} (Figure 2.9b, Table 2.8), corresponding with a difference in P load of 0.01 kg ha^{-1} yr^{-1}. Average BSE_I was 15% (Figure 2.9a, Table 2.8). The effect of BS was quite variable between seasons (Figure 2.9) and therefore BSE_{II} was statistically insignificant (Table 2.9: $P = 0.319$). Low $\overline{C}$ for P and BSE for Beltrum was expected, because no surface runoff towards ditch or reservoirs was observed, and groundwater level (Table 2.1) remained below the top soil layer with relatively high PSD (Table 2.2).

At Loon op Zand both BSE_I and REF-BS were low and variable for P_t (Figure 2.9, Table 2.8). Calculated difference in corresponding P load was only 0.04 kg $ha^{-1} yr^{-1}$. Although BSE_{II} was significant in the integrated REML analysis (Table 2.9, $P = 0.010$), the overall treatment effect (REF-BS 0.0227 g m^{-3}) was entirely caused by Winterswijk (REF-BS = 0.2312 g m^{-3}). Note, the interaction with location was also significant (LxT, $P = 0.031$). The effect of BS at Loon op Zand was most pronounced for the dissolved fractions P_{ts} and PO_4-P (Table 2.8), probably due to the very high PSD (Table 2.2; Koopmans et al., 2004). Correspondingly, REF-BS for $\overline{C}$ PO_4-P was highest for this site (0.044 g m^{-3}, Table 2.8). The P_t concentration in the ditch outside the reservoirs (not shown) was twice the value inside reservoirs during the whole experimental period, only at this site. This points to large P loads, also before the experiment, and therefore to a high P status of the ditch bottom. Possibly, P release from the reservoir bottom levelled out potential differences between BS and REF.

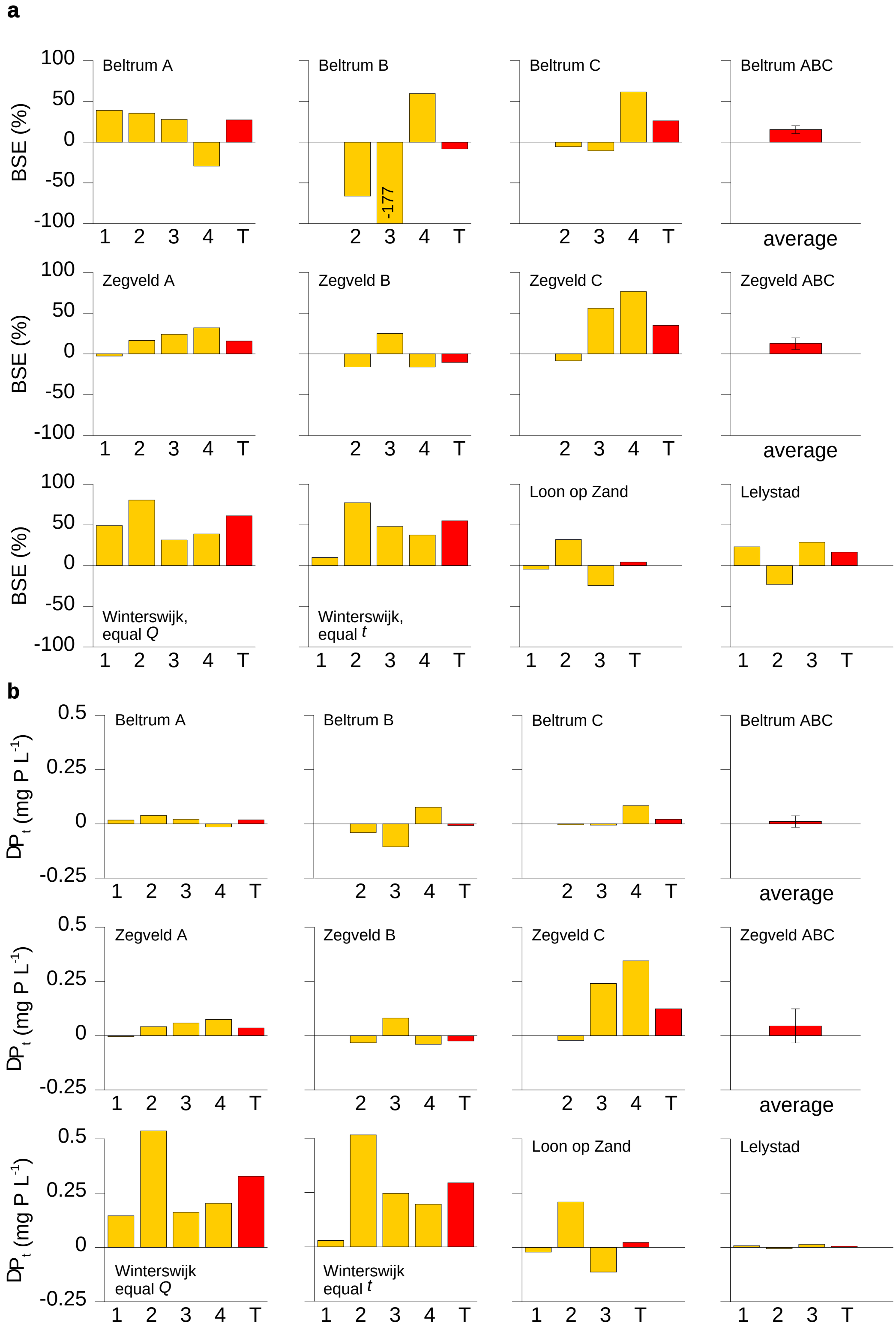


Figure 2.9

***a)** Buffer strip effectiveness (BSE_i, Eq. [2.3]) for flow-weighted average P_t and **b)** absolute difference in P_t between REF and BS (DP_t), both for individual leaching seasons (1,2,3,4) and for the total experimental period (T), for all locations and replicates, and for periods of equal discharge and for Winterswijk also for periods of equal time*

At Winterswijk we found an average BSE_I of 61% and REF-BS of 0.328 g m^{-3} P_t (Figure 2.9, Table 2.8), corresponding with an estimated difference in P load of 0.72 kg $ha^{-1}yr^{-1}$. Although this difference was mainly determined by particulate P, also P_{ts} and PO_4-P showed a clear treatment effect (Table 2.8). The significant treatment effect on P_t in the integrated statistical analysis (Table 2.9) was almost entirely caused by the effect in Winterswijk (BSE_{II} = 56.9%, REF-BS = 0.2312 g m^{-3} P_t) and the treatment effect at Winterswijk was consistent for all years (Figure 2.9). There was also a clear difference in net P withdrawal (Table 2.4) and Pw declined more in the BS than in the REF (Noij et al., submitted).

At Lelystad we found an average BSE_I of 4 % for P_t, while REF-BS < 0.01 g m^{-3} (Figure 2.9; Table 2.8 and Table 2.9). Estimated difference in P loads was only 0.02 kg $ha^{-1}yr^{-1}$. Although Pw, P-AL, C_{gw} and $\overline{C}$ were consistently lower in the BS compared to REF (Noij et al., submitted), a low BSE could be expected in Lelystad, because both PSD (Table 2.2) and groundwater level were low (Table 2.1). Low groundwater levels were due to the combination of tile drains and low ditch water level (Table 2.1). Tile drainage prevented shallow flow and surface runoff.

At Zegveld we found high $\overline{C}$ P_t and P_{ts} concentrations. Also particulate P was high (Table 2.8), probably because of surface runoff. Concentrations were about 50% lower compared to ditch concentrations reported by Van Beek (2007) for another site in *Holland peat*. These authors attributed 33-83% of P load to the eutrophic subsoil and reported very high C_{gw} below 50 cm bss (0-10 g m^{-3} P). We, however, found a very low PSD of 2 - 8% between 50 and 200 cm bss at Zegveld. Correspondingly, C_{gw} PO_4-P was negligible.

The statistical analysis on all locations revealed a significant effect for P_t (Table 2.9). This is attributed to the dominant effect observed for Winterswijk (BSE_{II} = 56.9%). No treatment effects were obtained for the Beltrum and Zegveld sites (Table 2.9). Despite substantial $\overline{C}$ the BS effect was low (Figure 2.9, Table 2.8) and not significant (Table 2.9). Average BSE_I was 18% and REF-BS was only 0.032 g m^{-3} for $\overline{C}$ P_t (Figure 2.9, Table 2.8) and less for P_{ts} and PO_4-P. The difference in $\overline{C}$ for P_t corresponds with a very small P load of 0.06 kg ha^{-1} yr^{-1}.

A small decline in the P status of the top soil (0-30 cm bss) underneath the BS treatment was observed for all mineral soils. However, also for the REF treatments a decline was observed, so that it is not yet possible to indicate a treatment effect based on the change in soil properties (Noij et al., in prep.).

2.3 Discussion and conclusions

2.3.1 Evaluation of the method

The experimental period (three or four leaching seasons) was sufficient to overcome hydrological time lag at all sites, because 50% of final tracer recovery was reached within one or two leaching seasons (Table 2.3). So water discharged from the REF and BS strips into the reservoirs must have been influenced by treatment and low BSE cannot be attributed to hydrological lag. We succeeded to recover the major part of *PS* in the reservoirs. Spatial differences in tracer recovery and discharge (*Q*/*PS*) between and within treatment pairs were likely caused by variation of soil and hydrological properties, including surface runoff at Zegveld (Holland peat). Rozemeijer et al. (2010b) and Van der Velde et al. (2010) also found substantial variability in discharge along the ditch with the same type of reservoirs. Discharge variability along the ditch is also well known from drain pipes. Hoogland et al. (2010) confirmed spatial variability of discharge by geostatistical modelling of measured groundwater levels along the ditch for the location Loon op Zand. As we used flow weighted concentration for periods of equal discharge for BSE assessment, we excluded an effect of discharge, except for Winterswijk. In Winterswijk both discharge and tracer recovery, and nutrient loads from the BS were much higher than from the REF treatment, due to a larger catchment area.

Thanks to the crop measurements we could confirm a relevant difference in net nutrient withdrawal between both treatment strips at all sites, so we can be sure that the BS treatment was reflected in the crop and the rooted soil layers. Nevertheless, lower net crop withdrawal was reflected in lower groundwater concentration at some sites only, and even less in lower reservoir concentration. At most sites low C_{gw} and $\overline{C}$ at BS were not caused by the treatment. We were able to detect this thanks to the REF treatment in the experimental set up. Dosskey (2002) and Heinen et al. (2012) already stressed the importance of including a REF treatment to avoid incorrect attribution of processes to the BS that may also occur without, like denitrification or mixing with deeper groundwater. Most authors checked or corrected for mixing in a situation without BS, but not for denitrification. Whether earlier assessments of BSE without REF were incorrect or not, depends on the research question. If introduction of BS on agricultural fields was evaluated, like in our study, the added value of BS compared to REF should have been established. However, if the effectiveness of a lower natural riparian zone next to upland agricultural area was evaluated, like in Ballestrini et al. (2008), Dhondt et al. (2002 and 2006), Hefting (2003) and Hoffmann et al. (2006), a REF is less relevant. Such research aims at quantifying the functioning of an already existing ecotone (Haag and Kaupenjohann, 2001). The Italian Po valley provides an intermediary case with traditional (i.e. existing) BS with trees and grass along agricultural fields (5-8 m wide). Reported high BSE for NO_3-N was assessed without REF and attributed to the hydrologic effect of the deeper tree roots and to denitrification (Balestrini et al., 2011; Borin and Bigon, 2002). We expect lower BSE values in case a REF would have been used, because denitrification would also occur without BS.

Reservoirs were valuable in our situation with unknown contribution from different flow paths, because they collected all flow paths. They revealed high spatial variability in discharge along ditches. Therefore, reservoir measurements before BS installation would further improve the experimental set up (Heinen et al., 2012).

In conclusion, experimental evaluation of BSE in agriculture should include a reference treatment. Reservoirs are recommended in case of unknown discharge flow paths.

2.3.2 Nitrogen

We found very low BSE for N at all five sites which are assumed to be characteristic for the Dutch lowland agriculture, much lower than BSE values reported for other circumstances (e.g., Barling and Moore, 1994; Dosskey, 2002; Mayer et al., 2005, 2007; Muscutt et al., 1993; Parkyn, 2004; Polyakov et al., 2005; Wenger, 1999). Only at Zegveld the BSE for N_t was statistically significant. At this site BSE_I was, however, low (10%, Table 2.6), especially compared to the relatively large proportional area of the BS (17%) on these narrow fields. At all sites low BSE can be explained by (i) N removal through denitrification in the soil of treatments and adjacent field, and (ii) by hydrologic factors (Table 2.10).

Denitrification

Although the non-fertilized BS reduced N surplus at all sites (Table 2.4), this was only reflected in lower C_{gw} below BS in Beltrum (deep sand) and Loon op Zand (interrupted sand) (Table 2.5). The corresponding BSE_{III} was 66% and 90%, respectively. We did not calculate BSE_{III} for Zegveld, Winterswijk and Lelystad, because low C_{gw} at BS could not be attributed to a treatment effect, as C_{gw} was also low at REF. Except for Beltrum, at all sites the difference in C_{gw} between REF and BS was smaller than between adjacent field and treatments (Table 2.5). Judged by a decreased NO_3-N/Cl ratio (Table 2.6), this was caused by denitrification in the soil between field and treatments (Table 2.10, note 2), except in Zegveld (Holland peat). At the peat site net production of nitrate occurred, causing even higher C_{gw} underneath the BS (Table 2.5, N_{ts}). Mineralization of abundant organic matter and subsequent denitrification have annihilated potential differences between REF and BS. Denitrification also played a role during transport between treatments and reservoirs (Table 2.10, note 1). At all mineral soil sites denitrification between REF and reservoir exceeded denitrification between BS and reservoir (Table 2.5), and therefore levelled out potential differences between both treatments.

Table 2.10
Explanatory factors for low buffer strip effectiveness for nitrogen in lowland agriculture

Factor	Beltrum	Loon op Zand	Winters wijk	Zegveld	Lelystad
Low contribution of affected shallow flow and surface runoff to discharge, i.e. high contribution of unaffected deep groundwater	X				
Downward seepage, not reaching the ditch	X	X			
Drain pipes by passing treatments					X
Low discharge		X			
Low residence time in treatments			X		
Unequal discharge (area) of treatments			X		X
Surface runoff away from ditch				X	
Denitrification between field and ditch	X[1]	X[2]	X[2]	X[1,3]	X[4]

1 predominantly between treatments and ditch (0 - 5 m)
2 predominantly between field and treatments (> 5 m from ditch)
3 contributes less to the explanation of low BSE because N_t is not dominated by nitrate.
4 contributes very little to the explanation of low BSE because of drain pipes

Hydrologic factors

In Beltrum (deep sand) about half of *PS* discharged via deeper groundwater (>7 m bss) and by-passed treatments and ditch (Table 2.10; Heinen et al., 2012). Reservoirs were filled with a mixture of affected shallow (1-2 m bss) and unaffected deeper groundwater (2-7 m bss), with relatively high nitrate and Cl concentration. Therefore, $\overline{C}$ for Cl was in between deep (>30 g m^{-3} Cl) and upper groundwater concentration below treatments and field (Table 2.5). At Loon op Zand (interrupted sand), only minor part of the *PS* of the field was recovered in the reservoirs, due to downward seepage.

At Winterswijk (shallow sand) low residence time in the treatments (0.03 year) hampered nitrate removal, especially during winter. Even if we apply a year round average first order denitrification rate for upper groundwater in the NL (1.84 yr^{-1}; Heinen et al., 2012), nitrate removal in a 5 m strip remains below 5.4% (100%*(1-exp(-1.84*0.03))). Hence, N load from treatments to reservoirs practically equalled incoming loads from the adjacent field, and negative BSE was caused by higher incoming N load in the BS. Higher N load at the BS could be largely explained by higher discharge (*Q*, Table 2.3). By consequence, residence time and denitrification between field and reservoirs were lower at BS, causing also higher $\overline{C}$ NO_3-N and NO_3-N/Cl for BS (Table 2.5). At Winterswijk a potential treatment effect below the narrow strips (5 m) could easily be outdone by spatial variability in the much larger discharge area of the adjacent field (5 - 80 m). Such spatial variability may relate to both N dynamics (N surplus à C_{gw} à $\overline{C}$) and discharge (*Q*).

At Lelystad (Holland clay), we did not expect any BSE because drain pipes by pass the treatment (Muscutt et al., 1993). Treatments (5 m) represented only ~3% of the water divide distance (150 m, Table 2.1). Even if BS reduced N leaching to upper groundwater by 100% BSE could not exceed 3%. Lower $\overline{C}$ at BS can be explained by dilution because the BS reservoir received ~10% more discharge (Table 2.3, see also $\overline{C}$ for Cl in Table 2.5).

High spatial variability of tracer recovery and *Q*/*PS* between treatment pairs in Zegveld is attributed to the observed erratic surface runoff pattern. During winter groundwater reached soil surface and pools appeared

(Appels et al., 2011), part of which flowed from the more elevated border of the ditch to the lower centre of the field. This surface runoff was not recovered in the reservoirs (Table 2.3, *Q*/*PS* < 1) and could have contributed to the relatively low BSE obtained for this location.

Upscaling
The effectiveness of BS for N proved to be controlled by site-specific factors (Table 2.10). Hence, BS would have to be tailor-made, but our results predict little perspective for effective BS application in lowland plains. At sites with pipe drains that cover 40% of agriculture in NL (Massop et al., 2000), BS are not effective, as suggested before by Muscutt et al. (1993). For the relatively uniform Holland peat area (12.8%) BSE proved to be low. As for the sandy soils, our results correspond with nitrate removal on flood plains differentiated by aquitard depth range (Hill, 1996: optimum 1 - 4 m bss). At deep sand (33.5%), the aquifer clearly runs too deep (>> 4 m bss) and at shallow sand (2.4%) too shallow (<1 m bss) for effective BS. Best perspectives are likely offered by interrupted sand (16.1%), if sufficient lateral groundwater flow occurs, which is not the case in 36% of this class due to downward seepage, like at our site Loon op Zand. At the remaining 64% BSE will depend on groundwater level and organic matter dynamics that control denitrification.

We expect higher BSE if surface runoff occurs (Mayer et al., 2005, 2007). Fast transport routes reduce levelling off differences between BS and REF. Although surface runoff played a minor role in our study, it certainly occurs on plain fields (Appels et al., 2011). However, BS would need to be specifically designed for effective abatement of surface runoff. Narrow BS with grass (< 5 m) could already prevent surface runoff of soil particles and spills of agrochemicals that easily occur if the agricultural land is utilized up to the very edge of the ditch. They further contribute to stabilization of the ditch bank and to biodiversity and could therefore be considered Good Agricultural Practice.

In conclusion, introduction of 5 m wide non-fertilized grass BS to reduce N loads from lowland agriculture to surface water is not effective. Low BSE is caused by site specific factors governing hydrology and denitrification in the soil between field and ditch. Best perspectives are likely offered within the hydrogeological class with interrupted sand (16.1% of NL), provided a number of additional conditions are fulfilled, such as sufficient lateral groundwater flow (64% of this class) and restricted denitrification, which depends on groundwater level and organic matter dynamics.

2.3.3 Phosphorus

High BSE for P (61%) was found on the site with very shallow flow and gentle slope (Winterswijk). Low and statistically insignificant BSE was found at all other sites. Even at high PSD in the top soil (Loon op Zand, Beltrum) BSE was low, because there was no shallow flow. Although shallow flow and surface runoff were substantial at the Holland peat site, low BSE for P at Zegveld could be attributed to the negligible effect of the BS on the P status of the peat soil. Evidently, shallow flow or surface runoff is a precondition for effective BS for P. According to international literature (Section 1.3), BSE for P is primarily determined by surface runoff and corresponding erosion. However, in our experiments we hardly observed surface runoff to the ditches, let alone erosion. As more surface runoff occurs, BSE for P will also increase in a plain delta with deeply permeable soils like the Netherlands, but will most probably never reach the level of sloping areas with impermeable subsoil, often found in other areas of Europe. On the other hand, in the Netherlands relatively more P leaching to groundwater occurs on so called P leaking soils, caused by former accumulation of P surpluses in the soil due to high animal density and intensive use of animal slurry.

The effect of accumulated net P withdrawal on P status of the mineral soils was still small after three or four years and negligible for the peat soil. Other research shows the difference in P status of the soil between REF and BS will increase and appear at greater depth with time (Van der Salm et al., 2009). Therefore our hypothesis that BSE increases with on-going net P withdrawal still stands. The time lag for BS to take effect increases with the buffer capacity, i.e. the ratio between adsorbed and dissolved P at equilibrium (Koopmans et al., 2004).

As there was only one mineral soil site with substantial shallow flow, we could not directly test our hypothesis that BSE for P is positively related to PSD. However, the highest difference in reservoir PO_4-P concentration between REF and BS at Loon op Zand (Table 2.8) corresponded with the highest PSD at this site (Table 2.3). This indicates potentially high BSE in situations with high PSD and more shallow flow than in Loon op Zand. More shallow flow can be expected on 64% of the hydrogeological class e) interrupted sand (Hoogland et al., 2010). Whereas we found negative BSE for N at Winterswijk (Section 2.3.2), we found a clear positive effect for P. We attribute this remarkable difference to the short travel time in the BS, on average 10 d. While such a small time period is insufficient for N retention in the BS during winter, it is likely enough for P adsorption (according to Koopmans et al. (2004) reaction time for anorganic P adsorption < 1 d). Winterswijk combined shallow flow with a relatively high PSD, and as such can be regarded a P leaking soil. We therefore expect high BSE for P at all P leaking soils, especially those with very shallow flow and P concentrated in the top soil layer.

In conclusion, shallow flow or surface runoff is a precondition for effective BS for P, also on plain well drained land. Our results further suggest that BSE for P is positively related to PSD and accumulated net P withdrawal from the BS. The time needed for BS to take effect further depends on the buffer capacity of the soil. Consequently, reduction of P loads from agricultural lowland with BS is only effective in specific areas with high surface runoff or shallow flow, in combination with high PSD, especially in the top soil.

3 Model study

Box 3.1 Abbreviations and symbols used in Chapter 3

Symbol	Units	Description
B	m	Width of field ditch
B_0	m	Buffer Strip length at x=0
BSE	dimensionless	Buffer Strip Effectiveness
$C(t)$	kg m^{-3}	Concentration as a function of time
$C(x)$	kg m^{-3}	Steady state concentration as a function of distance
C_0	kg m^{-3}	Input concentration at groundwater level
$C_{0,BS}$	kg m^{-3}	Input concentration to groundwater below the buffer strip
$C_{0,REF}$	kg m^{-3}	Input concentration to groundwater below the reference strip
c_0	d	Hydraulic resistance of ditch bottom
c_1	d	Hydraulic resistance of aquitard
E	dimensionless	Elasticity of the BSE with respect to a certain parameter
h	m	Thickness of top layer with relatively high microbiological activity
H	m	Depth of aquifer with constant thickness
H`	m	The effective thickness of the aquifer
H_0	m	Depth of aquifer at x=0
$h(x)$	m	Groundwater elevation as a function of distance
k	d^{-1}	First order decomposition rate coefficient
k_{11}	d^{-1}	First order reaction rate coefficient of biogeochemical domain (1,1)
k_{12}	d^{-1}	First order reaction rate coefficient of biogeochemical domain (1,2)
k_{21}	d^{-1}	First order reaction rate coefficient of biogeochemical domain (2,1)
k_{22}	d^{-1}	First order reaction rate coefficient of biogeochemical domain (2,2)
kD	m^2 d^{-1}	Transmissivity
L	m	Distance between water courses
L`	m	Field length corrected for the ratio between discharge and total precipitation surplus
$Load_{BS}$	kg ha^{-1} d^{-1}	Nutrient load to surface water for a field with buffer strip
$Load_{REF}$	kg ha^{-1} d^{-1}	Nutrient load to surface water for reference conditions
p	m	Surface water level
p_1	dimensionless	Linear increase coefficient of B_o with distance
p_2	dimensionless	Linear increase coefficient of H with distance
PAR		One of the parameters to be varied in the sensitivity analysis
Q	m^{-3} d^{-1}	Water discharge to field ditch
$Q(0)$	m^{-3} d^{-1}	Water flow rate at x=0
R	m d^{-1}	Recharge rate (precipitation surplus)
S	m d^{-1}	Upward seepage rate
$t(x)$	d	Time needed for travelling from distance x to the field ditch
$t_{1,1}(x)$, $t_{2,1}(x)$, $t_{1,2}(x)$, $t_{2,2}(x)$	d	Travel times in the 4 different biogeochemical domains
X	m	Distance from the centre of the field ditch
x_1, x_2	m	Positions relative to the centre of the field ditch
X_{BS}	m	Width of a buffer strip
X_R	m	Width of relatively wet zone adjacent to field ditch
ε	dimensionless	Porosity
φ	m	Hydraulic head below the bottom boundary
λ_L	m	Parameter composed of other hydrogeological constants
λ_B	m	Parameter composed of other hydrogeological constants

3.1 Modelling approach

Buffer strip effectiveness is expected to increase with time due to time lag of the processes involved, such as hydrological travel time, proceeding nutrient withdrawal from the harvested BS and organic matter dynamics. As our field study (Chapter 2) lasted for three or four years only, we wished to account for the time since installation of the BS in the modelling study. The tracer experiment of the field study showed hydrological lag time could be overcome after 1-2 leaching seasons. However, proceeding nutrient withdrawal without fertilizers and subsequent declining N and P status of the soil is expected to increase BSE for a longer time period. Therefore it was regarded necessary to apply a more complex dynamic modelling approach to assess the evolution of BSE until steady state, i.e. where BSE does not increase anymore with time. This approach was applied for N and P in Zegveld, and for N only in Beltrum, because measured P concentrations were too low there for calibration purposes (in other words P was not relevant in Beltrum).

The field study (Chapter 2) yielded estimates for buffer strip effectiveness (BSE) at five locations characteristic for the most important hydrogeological classes in the Netherlands. Such a field study is too laborious and costly to repeat for varying site specific conditions. Therefore, the modelling study should enable the interpolation and extrapolation of BSE-values to other locations, or the assessment of BSE for other locations with only a limited number of data available. In order to reach this goal it was necessary to consider the dominant spatially distributed key factors controlling the nutrient load from non-fertilized strips.

Applying a complex dynamic modelling approach to assess BSE for other locations was not considered feasible as this would require too much site specific data and calibration. Instead we developed a simpler analytical model for N for steady state situations, i.e. for the final BSE after a longer period of time. As this analytical model could not handle the much faster P retention processes in soil and as the very low measured P concentrations impeded calibration and validation of any model at most experimental locations, we did not extrapolate BSE for P by modelling. Instead we derived the following line of thoughts from the field study and other research for extrapolation purposes (Chapter 2). First of all, shallow flow or surface runoff is always a precondition for effective BS for P. As adsorption of P is completed very fast (<1 d), compared to decomposition of organic substances and nitrate (order < 1 yr), the travel time of draining water is irrelevant for P, as opposed to N. The inorganic P load is determined instead by the P status of the soil strata, adjacent to the ditch, through which the water is transported, and BSE therefore depends on the difference in soil P status of these layers between BS and REF (or rest of the field). The difference in P-status between BS and REF depends on the time since installation, because the BS continues to withdraw P from the soil, whereas P withdrawal from the soil in the REF is compensated for by fertilization. The difference in P-status between BS and REF, and thus BSE, further increases with high original P status and low buffer capacity of the soil.

The field data could not be used directly to test the analytical model for N, as the available field data did not refer to steady state situations. Therefore we applied the following procedure. We calibrated the complex dynamic model with the field data of two of our experimental sites, Beltrum and Zegveld. Then the dynamic model was used for long term calculations to achieve steady state. These steady state results were in turn used to calibrate the steady state analytical model. The calibrated analytical model was subsequently used to explore which key factors determine BSE (sensitivity analysis). Finally, these key factors were used to estimate the expected range of BSE for the NL.

This Chapter starts with a brief description of the dynamic 2D nutrient model that was developed to describe water and nutrient transport in a transect perpendicular to the ditch, and to calculate time dependent BSE (Section 3.2.1 - 3.2.3). The calibration of this model based on the field data collected at Beltrum and Zegveld and the results for the evolution BSE with time can be found in Sections 3.2.4 and 3.2.5, respectively. In Section 3.3 the principles of the analytical model for N are shortly described, together with its calibration on the dynamic model results. Finally Section 3.4 presents the results of the sensitivity analysis and the extrapolation of BSE.

3.2 Mechanistic dynamic models for field scale BSE assessment

3.2.1 Soil moisture and water flow: FUSSIM2

The FUSSIM2 model (Heinen, 1997, 2001; Heinen and De Willigen, 1998, 2001) describes two dimensional water movement, solute transport, and root uptake of water and nutrients in a soil transect perpendicular to the drain (Figure 3.1). For the discretization the flow domain is divided in layers and columns. Both vertical gradients and gradients perpendicular to the ditch can be simulated. Water movement is described by the state-of-the-art Richards equation, which is numerically solved for given initial and boundary conditions. The hydraulic properties of the soil are described by the widely used Van Genuchten (1980) and Mualem (1976) relationships, while hysteresis is described according to Mualem (1984). Root uptake of water and nutrients is described by models of De Willigen and Van Noordwijk (1987; 1994a,b). Alternatively, the uptake concept of Feddes et al. (1978) can be used, as was done in this study. The solute transport part of FUSSIM2 is disabled in this study, as we will use the more detailed solute transport model ANIMO (Section 3.2.2).

In this study FUSSIM2 is used to describe water movement through the soil towards the ditch. It simulates pathways of infiltrating water the dynamics groundwater levels, and the discharge from soil to ditch. Simulation results of water flow between columns and towards the ditch will be used as input for the ANIMO simulation (Section 3.2.2).

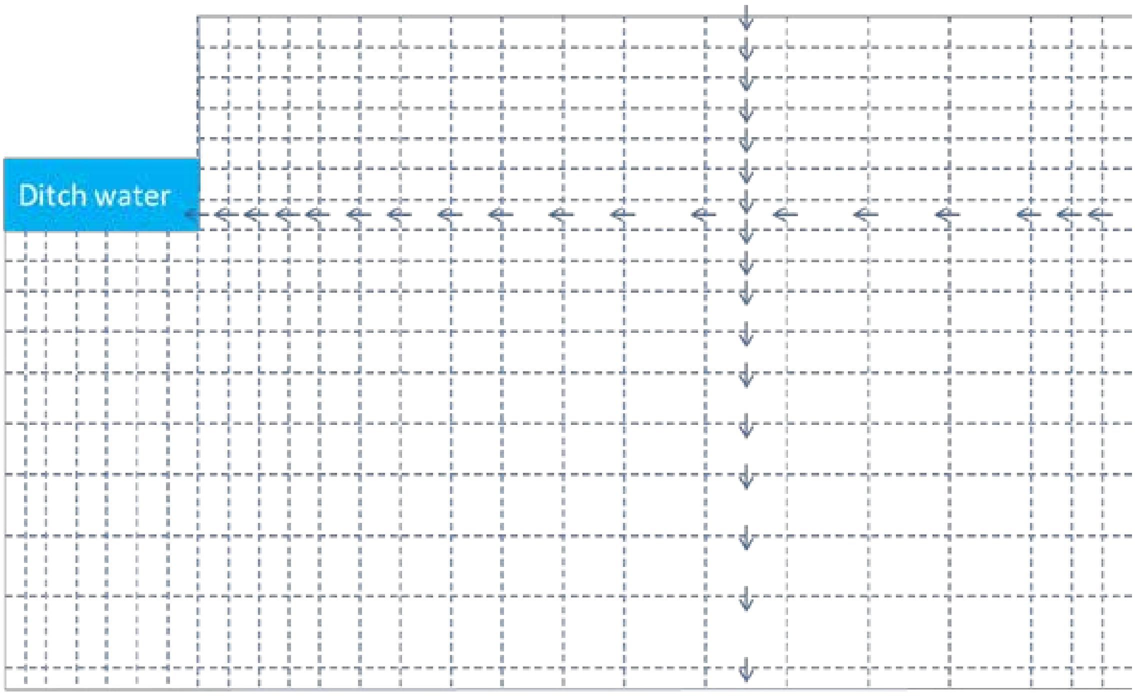


Figure 3.1
Soil transect with a the 2D grid with soil layers and columns, and a rectangular ditch. Each cell represents a FUSSIM2 calculation unit. Between all neighbouring units water fluxes are dynamically computed. The arrows indicate examples of local fluxes between units

3.2.2 Nutrient dynamics and leaching: ANIMO

The ANIMO model (Groenendijk et al., 2005) quantifies the relation between fertiliser application rate, soil management and the leaching of nitrogen (N) and phosphorus (P) to groundwater and surface water systems (Figure 3.2). The upper boundary of the model is the agricultural land surface, where nutrients are applied, the side boundary is the edge of the field, where N and P leach from soil to ditch. The lower boundary is defined at a hydrological boundary in the groundwater. ANIMO simulates nutrient transport to surface waters, but does not consider the transformation and transport processes within the surface water system itself.

ANIMO includes complete descriptions of the organic matter, nitrogen and phosphors cycle since these cycles are interrelated in farming systems and in soil biochemistry.

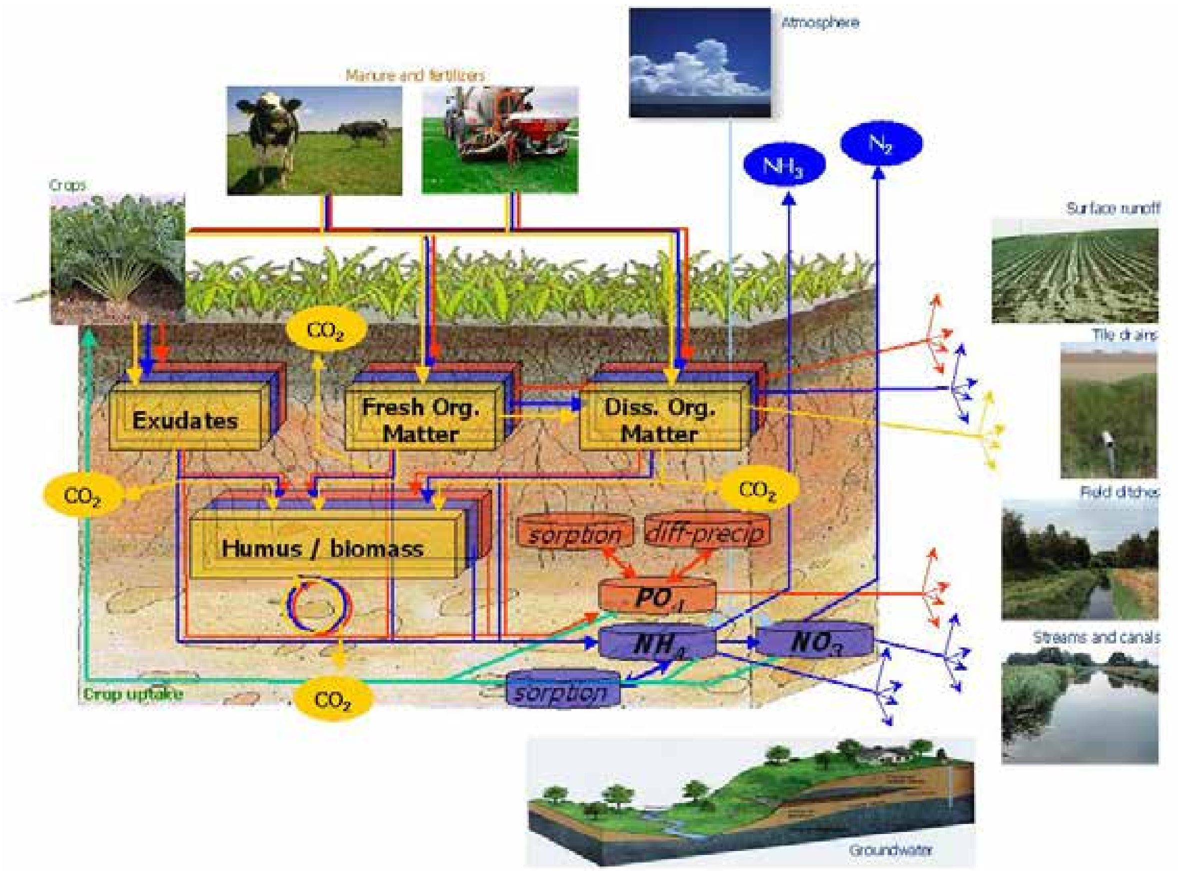


Figure 3.2
Schematic overview of processes simulated by the ANIMO. Arrows indicate the conversions and boxes the pools of organic matter (C: yellow), nitrogen (N: blue) and phosphorus (P: red)

ANIMO simulates the following processes:
- additions (fertiliser, manure, crop residues, atmospheric deposition);
- decomposition of recently added and native soil organic matter and related mineralization of N, P, and C;
- nitrification of NH_4 and denitrification of NO_3;
- NH_3-volatilisation and emission of CO_2, N_2, N_2O;

- instantaneous and time dependent sorption, and chemical precipitation of phosphates (Schoumans and Groenendijk, 2000);
- soil P status indices for Dutch fertiliser recommendations Pw on arable soils and P-AL on grassland;
- nutrient uptake by the crop;
- transport of dissolved nutrients and organic matter with the water flow ;
- surface runoff of water with dissolved nutrients and organic matter.

Being a 1D model, ANIMO describes fluxes in the vertical soil profile only, although the stand-alone version also predicts discharge to the different types of drainage systems: surface runoff and shallow trenches, tile drains, field ditches and canals. However, in this study we will couple the ANIMO model to the 2D FUSSIM2 model (described above). Hence, we will only use the 1D vertical nutrient transport processes of ANIMO. Lateral outflow from ANIMO will be used to feed neighbouring ANIMO models (see below).

The ANIMO model is part of the National Dutch modelling system STONE for the evaluation of fertiliser policy measures (Wolf et al., 2003). The ANIMO model has been reviewed and compared with other European models for several aspects, such as the organic matter and N cycle (Wu and McGechan, 1998), and the P cycle (Lewis and McGechan, 2002).

3.2.3 Integrated model: FUSSIM2-ANIMO

The FUSSIM2 and ANIMO models have been coupled via a shell (Rappoldt et al., 2008). Instead of transforming the 1D ANIMO model into a 2D version, we decided to keep the existing ANIMO model intact. ANIMO was coupled to FUSSIM2 as a soil column model, called 'instance' (Figure 3.3). The ANIMO instances are put side by side to match the grid for which FUSSIM2 delivers the hydrology (Figure 3.1). The horizontal arrows in Figure 3.3 indicate how water flux information of a certain soil layer is used by each ANIMO instance as a boundary condition for that layer. Of course, the vertical water fluxes between the layers were also exchanged between FUSSIM2 and ANIMO. The shell stores, exchanges and keeps track of information of the individual ANIMO instances.

FUSSIM2 and ANIMO were run sequentially. Potential feedbacks such as the influence of the soil nutrient status on crop development and evapotranspiration were not considered. The integrated model was run first with the most CPU-time consuming model FUSSIM2, for each separate hydrological situation, while storing the hydrological output data on a daily basis. Then later, the integrated model was run with ANIMO while using the stored FUSSIM2 output data as input for ANIMO. In this way, several nutrient scenarios can be run for the same hydrological conditions.

Special attention was paid to the order in which ANIMO instances had to be considered within each time step. Based on the main flow direction the most upstream ANIMO instance was run first, and then the next instances following the flow direction (Figure 3.3). For example, during a period with excess rainfall and outflow into the ditch the ANIMO instances were run from right to left in Figure 3.3.

We first determined if discharges and effluent concentrations for a rectangular soil-ditch interface (Figure 3.3) sufficiently resembled discharges and effluent concentrations for a true representation of a sloping ditch bank. Discharge proved nearly identical, and solute concentrations in the vicinity of the ditch were similar. Therefore, we modelled the system as a rectangular ditch.

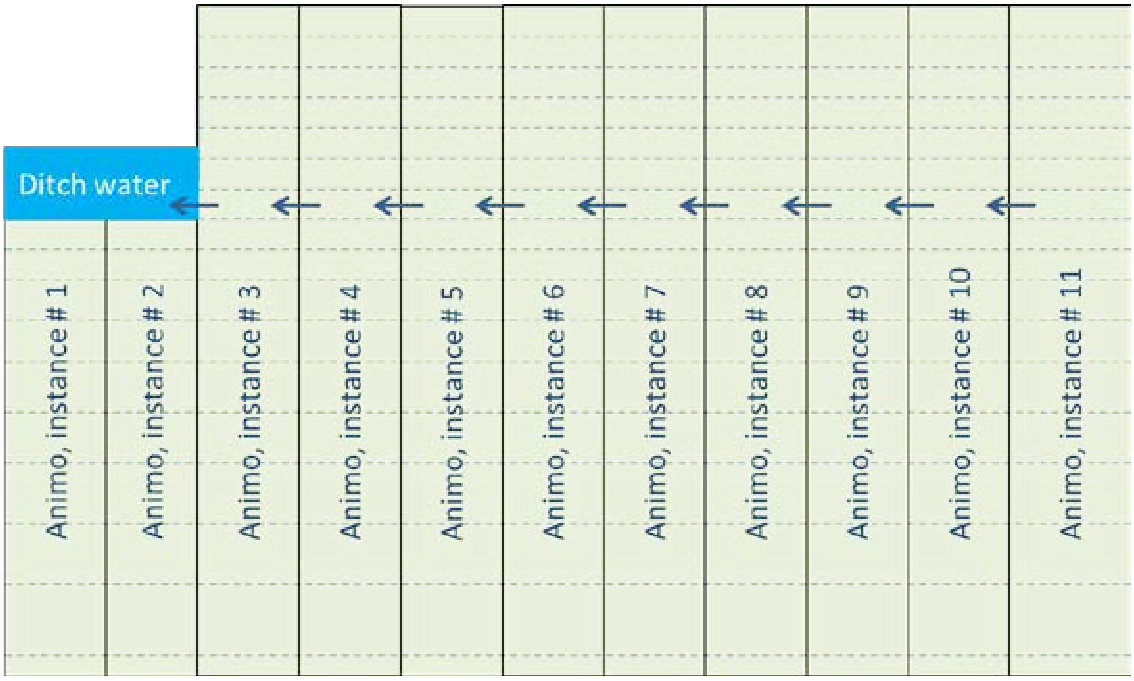


Figure 3.3

Similar soil transect with a the 2D grid with soil layers and columns as depicted in Figure 3.1. Each column represents a 1D ANIMO instance, which contains a whole number of FUSSMI2 calculation units. The horizontal arrows indicate how water flux information is used as input information and how solute concentrations are used as boundary conditions by each ANIMO instance

3.2.4 Application to Beltrum

Hydrology

The delineation of flow domain of the transect at Beltrum (Figure 3.4) was based on observed groundwater levels and on observed water discharges to the field ditch. The side boundaries of the flow domain for the integrated model were chosen at the centre of the field ditch (Figure 3.4, West) and at the water divide at 60 m from the ditch centre (Figure 3.4, East). The water divide of the local flow system was established based on the maximum elevation of the observed groundwater plane. From the observed groundwater elevations it appeared that the water divided is not constant with time, but deviates from season to season. The average position of the local maximum of groundwater levels was chosen as a boundary in our model. From the measured discharges it appeared that only a part of the area between field ditch and the water divide contributes to the discharge. The precipitation surplus of the other part of the field is conveyed to the deeper groundwater system and becomes a part of the regional groundwater flow. The ground surface is taken as the top boundary and an aquitard at 20 m depth is considered to be the bottom boundary. The assignment of soil physical parameters was based on the results of field investigations by (Heinen en Van Kekem, 2011; Table 3.1). Crop parameters, both for the normally managed maize crop and for the non-fertilized grass were adopted from STONE (Wolf et al., 2003). Since 2006 we accounted for the grass sown in after the maize harvest as a winter catch crop.

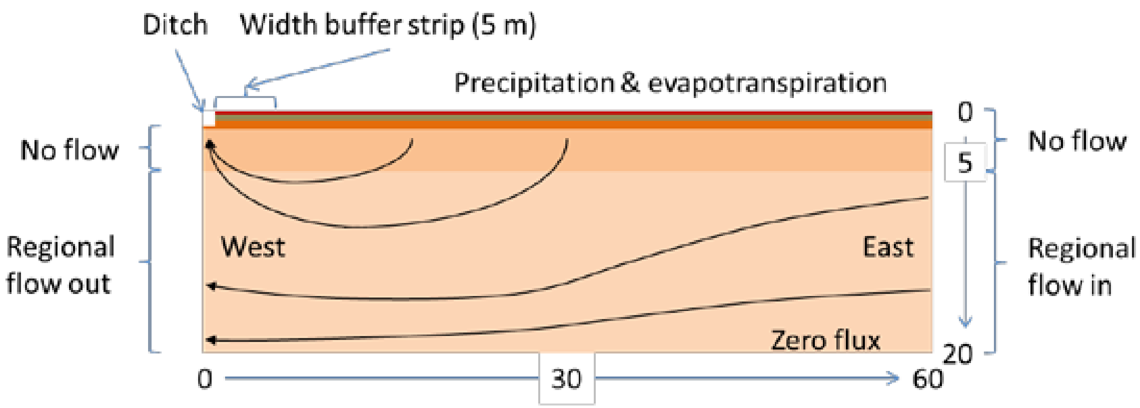


Figure 3.4
Flow domain of the transect at Beltrum with five different soil layers (Table 3.1), indicated by different colors, and the boundary conditions of the integrated model. At depth 20 m below soil surface there is an impermeable base; the horizontal distance refers to the center of the field ditch. Right boundary is determined by the water divide at 60 m from the centre of the ditch according to the groundwater elevation recordings

Table 3.1
Mualem-Van Genuchten parameters (REF's) assigned to the 5 soil layers of the Beltrum transect

Depth (cm)	θ_s	θ_r	α_d	α_w	n	K_s	λ
0 – 30	0.50	0.068	0.0144	0.0288	1.724	84.20[a]	1.724
30 – 80	0.28	0.039	0.0124	0.0248	3.511	35.70[a]	3.511
80 – 150	0.31	0.022	0.0112	0.0224	4.124	54.72[a]	4.124
150 – 500	0.31	0.022	0.0112	0.0224	4.124	50.0	4.124
500 – 2000	0.31	0.022	0.0112	0.0224	4.124	50.0[b]	4.124

a) Lower conductivity values assigned to the boundary layer in the vicinity of the field ditch (5% of K_s).
b) K_s refers to vertical conductivity value. Horizontal conductivity in layer 500-2000 cm was taken three times larger.

Ad 1 regional groundwater flow
The lateral boundaries were open boundaries for which daily water fluxes were derived to describe the dependency of groundwater pathways on the regional groundwater flow. The input at the Eastern side (Figure3.4) was derived from a relation between observed groundwater levels and the slope of the groundwater elevation during periods without discharge to the field drain in summer time. A daily time series for the inflow during our research period was calculated as the product of groundwater elevation slope and transmissivity.

Ad 2 groundwater levels - discharge relation
Additionally, daily water balances for the entire transect were calculated with the 1D SWAP model (Kroes et al., 2008), with equal meteorological data, crop characteristics and soil physical properties to derive time series for the part of the precipitation surplus which becomes a part of the regional flow to be used as a boundary condition for FUSSIM2. After calibration of the SWAP model on groundwater levels and drain discharges, the water fluxes at the bottom boundary were considered to be the net recharge to the regional groundwater system. These were added to the estimated inflow of regional groundwater at the Eastern boundary to obtain values for outflow fluxes at the Western boundary (Figure 3.4).

The calibration of the FUSSIM2 model was performed by comparison of calculated and observed groundwater levels after variation of a) saturated conductivity values of layer 4 and 5, b) the anisotropy factor of all saturated conductivity values, and c) a multiplication factor for the lateral inflow time series as derived from the procedure above (ad 1 and ad 2). The results were inspected visually and judged satisfactory after a series of model trials (Figure 3.5). The application of a formal calibration method would have been too time consuming with limited added value for the purpose of our study.

The cumulative water discharges to the field ditch are compared with measured values in Figure 3.6. The results of the different replicates are depicted as well. This gives an impression of the variability of the measured water discharges to the field ditch.

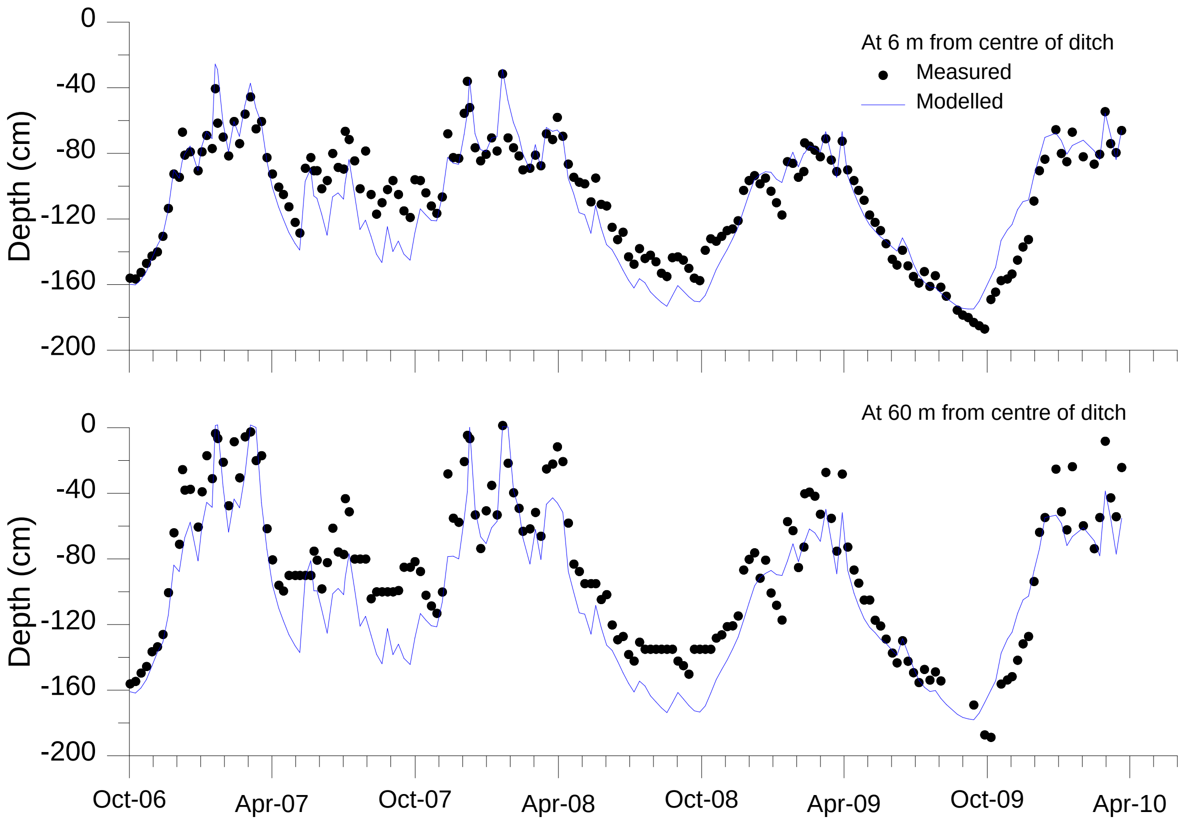


Figure 3.5
Simulated (FUSSIM2) and observed groundwater levels at 6 and 60 m (water divide) from the centre of the field ditch (Figure 3.4)

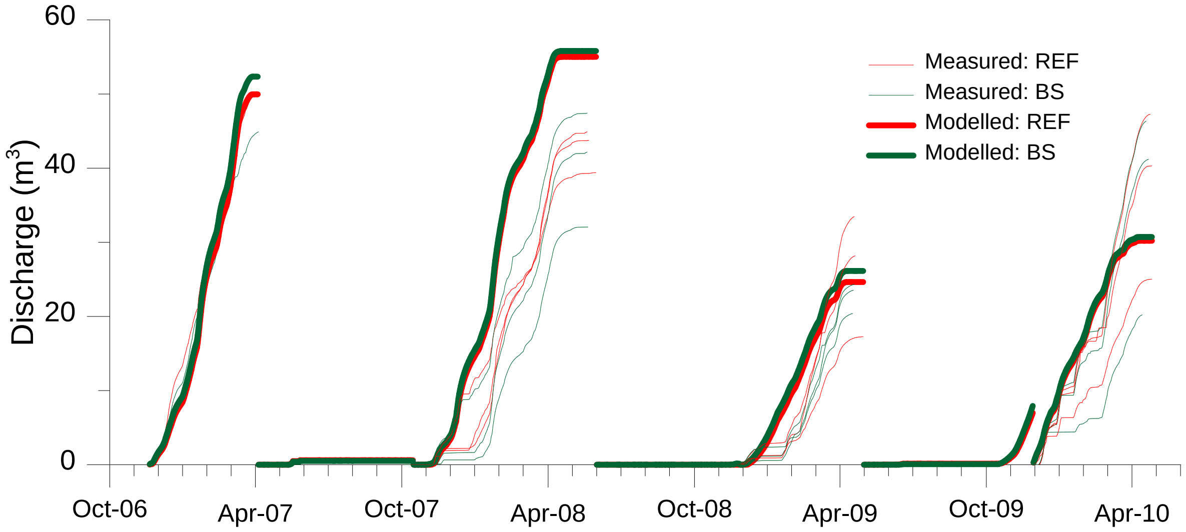


Figure 3.6
Simulated (FUSSIM2) and observed discharges in different replicates at Beltrum

Since precipitation rate is equal for both reference (REF) and BS, and the land surface of local flow system is mainly covered by maize for both treatments, we do not expect much difference in discharge due to treatment. In our calculations discharge from the BS is slightly higher, which can be explained by lower evapotranspiration rate of the non-fertilized grass strip compared to maize. The simulated discharge coincides well with the observed discharge for three of the four research years. In 2007 the model overestimated discharge by 33%, compared to the observed average. The procedure to derive inflow fluxes of regional groundwater at the Eastern side assumed an equal seasonal tendency, corrected for meteorological conditions, and did not account for possible changes in the regional groundwater flow due to adjustments of eg. pumping rates of drinking water wells and wells for building constructions and the possible adjustment of the water levels of larger streams at a greater distance. In the autumn of 2009 the simulated discharge period started somewhat earlier than the observed one.

Although the fits between simulated and observed groundwater levels and simulated and observed discharge rates were not perfect, the results were considered sufficiently satisfactory to proceed with the nutrient leaching modelling and the analysis of BSE.

Nutrient leaching

Farmer records were used for a time series of animal and industrial fertilizer applications in the model input. Nutrient withdrawal by the crop was measured on a seasonal basis (Noij et al., 2012 for N; and Noij et al. in prep. for P). Soil chemical parameters were taken from soil sampling results (Table 3.2).

Table 3.2
Soil input data for ANIMO for Beltrum (Heinen and Van Kekem, 2011)

Depth (m)	OM (% by weight)	Dry bulk density (kg m^{-3})	C/N-ratio
0 – 0.3	5.7	1174	14.9
0.3 – 0.8	3.9	1712	14.9
0.8 – 1.5	2.1	1712	12
1.5 – 5	0.7	1687	16
5 – 20	0.7	1687	16

Initialisation runs were performed to establish initial N and P concentrations in, and stocks of organic matter. We assumed thirty years of grassland land use before 2000, and maize from 2001 onwards, according to farmers information. Fertilizer and crop uptake rates were taken from the corresponding period and Eastern sand region of the STONE model. For the calibration of the ANIMO model we followed a stepwise procedure. The first step was for maize cultivation under average field conditions based on water fluxes simulated by the 1D vertical SWAP model. In the second step, the water fluxes and soil water contents of the FUSSIM2 model were used. The model was calibrated with observed nitrate concentrations in soil moisture and groundwater at different depths by varying the following variables: the mineralisation rate constant and the N weight content of the stable humus/biomass pool, half saturation value of the Monod nitrate response function for denitrification and the first order rate constant for denitrification in the case of nitrate limited denitrification rates. After a number of trials, the results of the ANIMO model were judged satisfactory by visual inspection (Figure 3.7) for studying the long term impacts of a buffer strip and effect of BS width.

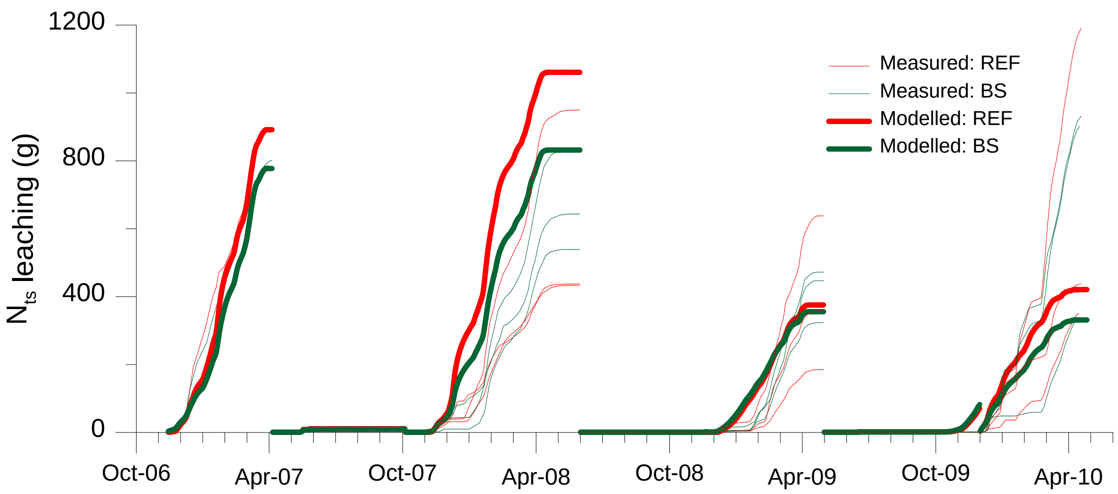


Figure 3.7
Simulated and observed cumulative N_{ts}-loads to the field ditch in Beltrum

Calculated BSE

The calibrated FUSSIM2-ANIMO model was run for 60 years with different BS widths. Calibration was conducted for the whole field research period, but extrapolation to long term steady state conditions with newly established equilibria was done with three representative meteorological years. Hydrology of 2007, 2008 and 2009 can be considered as representative for average climatic conditions, with 2007 as a relatively wet year, 2008 as a relatively dry year, and 2009 as an average year. This three year cycle was replicated 20 times with equal fertilizer and crop uptake rates. Figure 3.8 shows the evolution in time of BSE for total N for different BS widths. A near steady-state condition is achieved after 30 to 60 years for BS widths less than or equal to 19 m. For nitrate nearly identical results were obtained, as nitrate was the dominant component in total N. During first years BSE fluctuates due to the differences in weather between 2007-2009. This fluctuation levels off as time progresses. Final BSE-values have been calculated both for total N and nitrate loads (Figure 3.9). The final BSE for a 5 m strip width is 19.5% for nitrate and 18.5% for total N. If the entire field (60 m) would have been managed like a buffer strip, a maximum BSE of 70 - 75% would have been reached (Figure 3.9). A BSE of 100% is not possible due to so called background loads. In Beltrum three types of background sources can be distinguished:

1. atmospheric deposition;
2. slow release from stable organic matter pools;
3. lateral influx from the regional groundwater system (Figure 3.4).

We expect the latter source to be insignificant, because the flow between the local and regional system in Beltrum is mainly in downward. We can now calculate the fertilizer effect of the BS (hypothesis in Section 1.3). The areal fraction of the BS in Beltrum is 5/60 = 8.3%. Based on the ultimate BSE of 70-75% for a 60 m BS, the expected N load from an non-fertilized strip will be 25-30%. Hence we can expect a final N load from a 60 m field with a 5 m BS of (5*30%+55*100%)/60=94.2%. That is a fertilizer effect of 5.8%, much less than the BSE of 18.5% derived above. Apparently there **is** an additional specific BS effect (see hypothesis in Section 1.3).

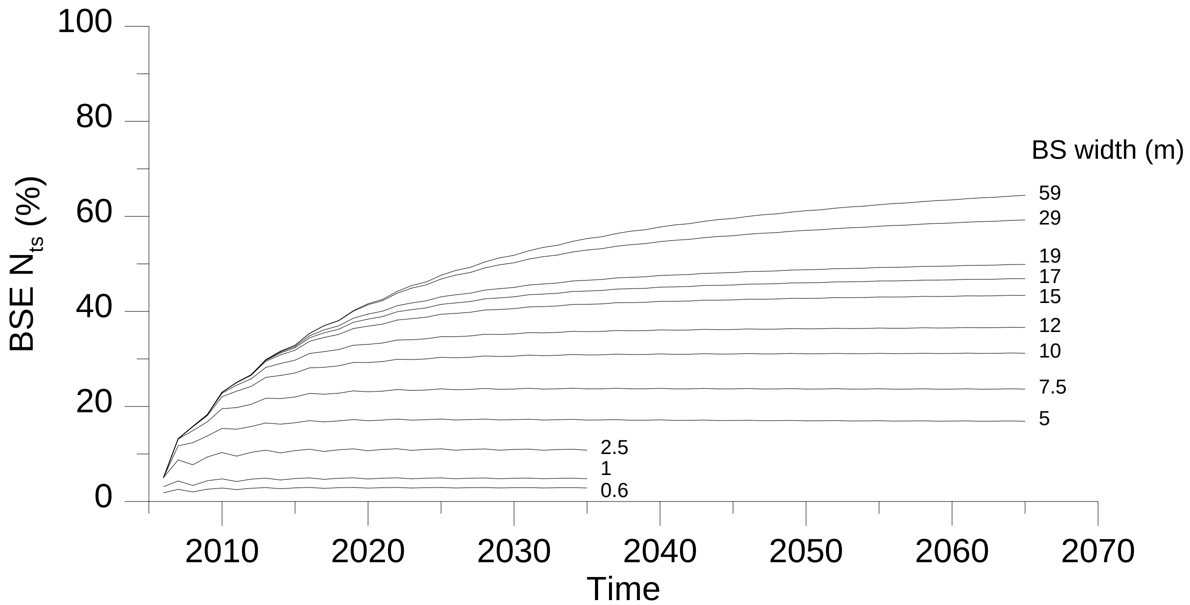


Figure 3.8
Buffer strip effectiveness for N_{ts} at Beltrum as a function of time for twelve buffer strip widths (m) calculated with the FUSSMI2-ANIMO model. Observed phosphorus concentrations (Chapter 2) were too low to justify model calculations.

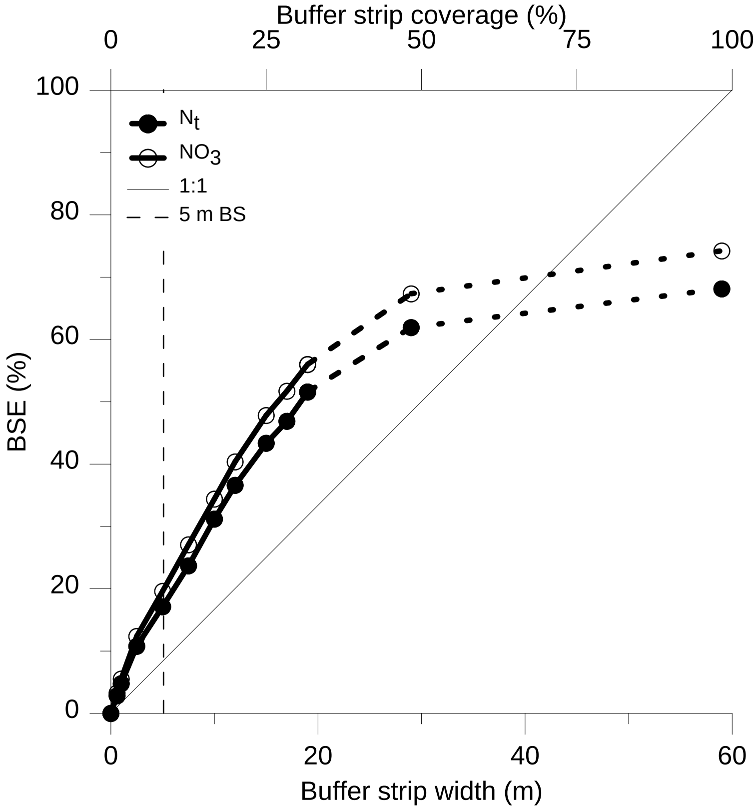


Figure 3.9
Final buffer strip effectiveness for nitrogen at Beltrum, derived from long term model results of FUSSIM2-ANIMO

3.2.5 Application to Zegveld

Hydrology

The Zegveld experimental field is located between two ditches (Figure 3.10). The flow domain consists of a 5.2 m thick peat layer on top of a relatively thin but low permeable clay layer. This clay layer forms the boundary between the peat layer and the deep sandy aquifer. We only considered flow in the peat layer. A detailed regional groundwater flow model was available for the region in which Zegveld is located (Jansen et al., 2007). The vertical exchange flux between the deep sandy aquifer and the peat and peaty/clay top system was extracted from the results of this regional model and was used as bottom boundary flux condition for FUSSIM2. We assumed symmetrical conditions on either side of the ditches, i.e. no lateral flow could be used as side boundary condition for the centre of the ditches (Figure 3.10). The soil physical parameters of the soil layers were taken from Heinen and Van Kekem (2011; Table 3.3). The eastern ditch was 2 m wide and approximately 80 cm deep and replicate A was located in this ditch. The other ditch, for replicates B and C, was located at the western side and was 3 m wide and approximately 90 cm deep. The water level was maintained at approximately 60 cm below soil surface.

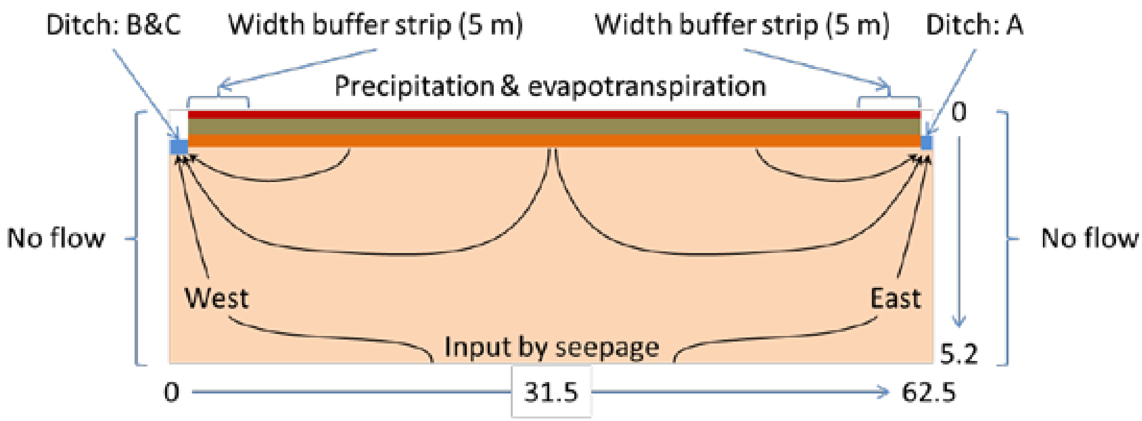


Figure 3.10
*Flow domain of the transect at the Zegveld experimental field with four different soil layers (Table 3.3), indicated by different colors, and the boundary conditions of the integrated model. The peat layer reaches to a depth of 5.2 m below soil surface where input by seepage occurs from the clay layer below; the horizontal distance runs between the two centers of the field ditches. Due to symmetry the lateral boundaries are no-flow boundaries, and the assumed water divide is located the middle. The vertical scale is stretched by a factor 4 (x-scale = 4*y-scale)*

Table 3.3
Soil physical characteristics of the four soil layers in the Zegveld profile

Depth (cm)	θ_s	θ_r	α_d (cm^{-1})	α_w (cm^{-1})	n	K_s (cm/d)	λ
0 - 15	0.7839	0.3451	0.0811	0.1622	1.2231	30.0	0.0001
15 - 50	0.8724	0.0	0.0091	0.0181	1.2118	30.0	0.0001
50 - 75	0.9219	0.0	0.0177	0.0354	1.3043	30.0	0.0001
75 - 520	0.9243	0.0	0.0219	0.0438	1.3384	30.0	0.0001

The measured daily average ditch water levels were imposed to the model. From these levels the fixed pressure head at the ditch bottom and the ditch wall were computed and were used as local boundary conditions.

We did not model crop growth and development. Instead we used a constant leaf area index of 3 and a constant rooting depth of 25 cm for both REF and BS grass. The rooting depth of 25 cm appears to be plausible for the relatively wet conditions of peat soils.

Groundwater levels at 3 m from the centre of both ditches and at the middle of the field were simulated relatively well (Figure 3.11). Note the reference point of the measurements (top of groundwater tube) also changed in time due to swelling and shrinking of the peat (order 2 - 6 mm/yr). Modelling this dynamics would require a very complex model and was therefore neglected. Instead, we used the average soil surface level as reference. This might partly explain the relatively high simulated ponding levels at the middle of the field during wet periods.

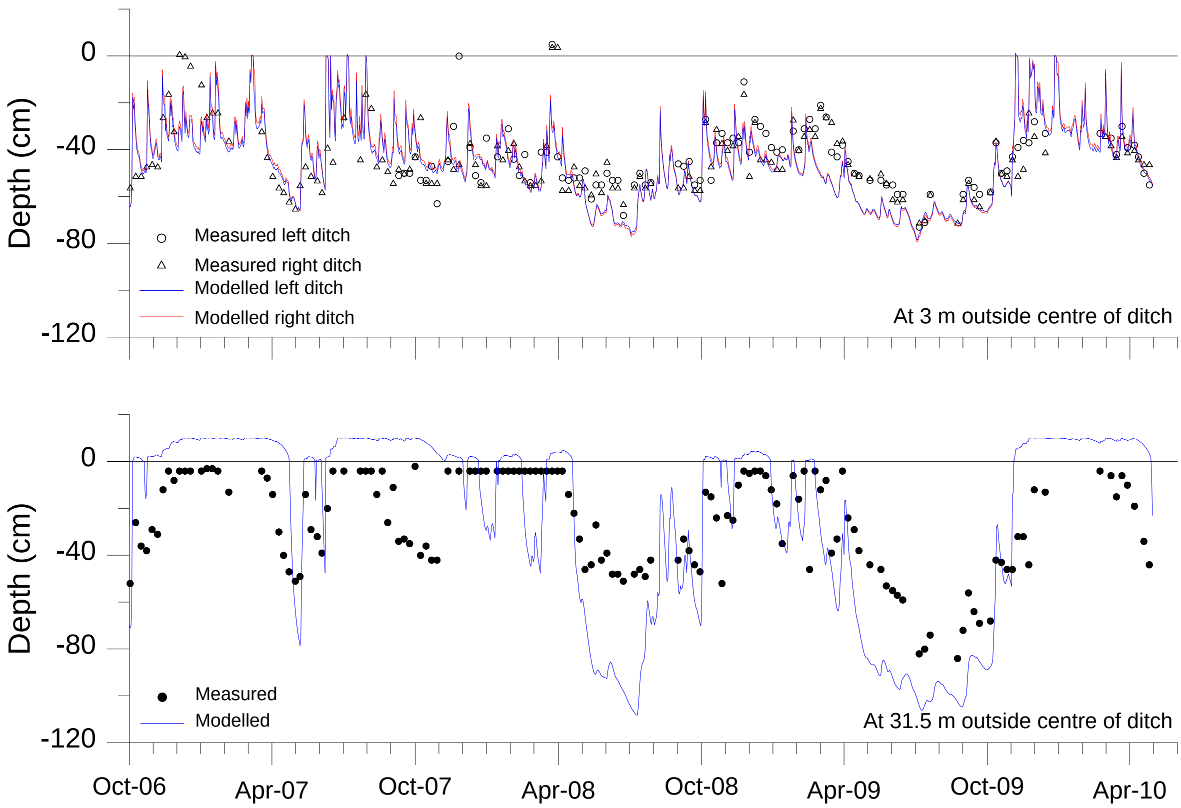


Figure 3.11
Simulated and measured groundwater levels at 3 m from the ditch (above) and at the centre of the field (below) at Zegveld

The model underestimated averaged total discharge of water to both ditches during the first leaching season (Figure 3.12). This could be partly attributed to surface runoff: adding simulated surface runoff to discharge results in closer correspondence with the measured discharge. No surface runoff was simulated for the second and third leaching seasons, when simulated discharge was comparable with the average discharge into the six reservoirs. For the last leaching season simulated discharge was somewhat higher than measured, even without surface runoff. This might be due to a long frost period, which could not be taken into account in the model.

The data of Figure 3.12 refer to the gross discharge, i.e. without subtracting infiltration during summer. The difference between simulated gross and net discharge indicates the level of infiltration at Zegveld. Infiltration was also observed in measured data (not shown).

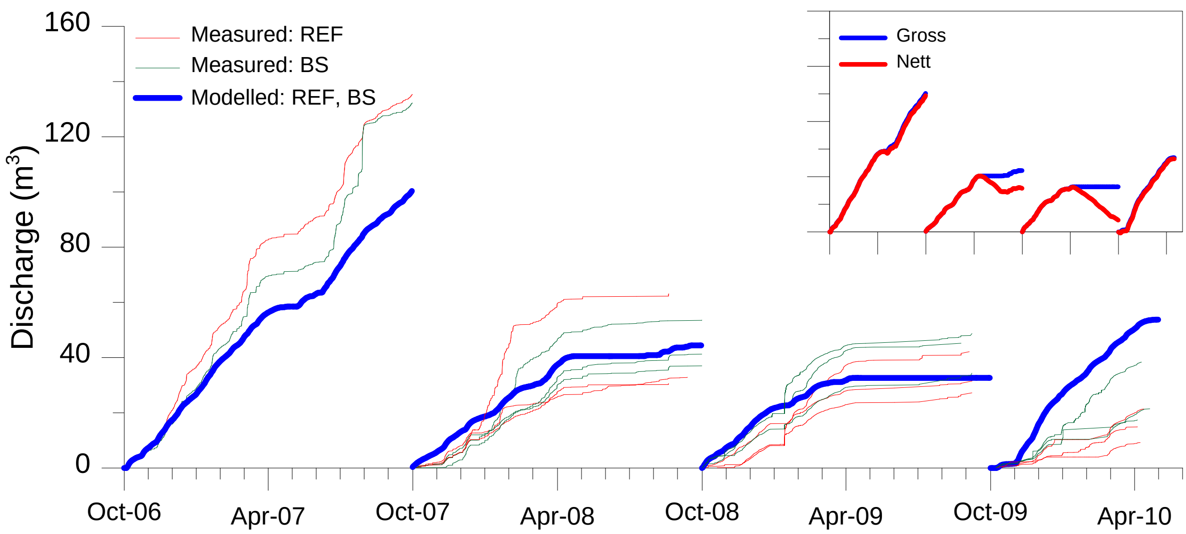


Figure 3.12

Simulated and measured accumulated discharge in Zegveld. In-set shows the difference between gross and nett discharge for the four seasons

Although the fits between simulated and observed groundwater levels and simulated and observed discharge rates were not perfect, the results were considered sufficiently satisfactory to proceed with the nutrient leaching modelling and the analysis of BSE.

Nutrient leaching

Grassland management and fertilizer rates for the experimental period (2006-2010) were taken from the records of the experimental farm to which this field belonged. Manure was applied to the reference strips five times and industrial fertilizer 19 times. Grassland was used for combined grazing and cutting according to normal practices (on average six times per year). Sheep (1.0-1.3 livestock units per hectare) were allowed to graze the field (including BS), for 89 days during the winter periods of 2006-2008, but not for the winter periods in 2009 and 2010. Estimated excretion was nutrient input for the model. Input data on soil properties for ANIMO were taken from Heinen and Van Kekem (2011, Table 3.4). In case of infiltration of ditch water in the summer season, the concentrations of the infiltrating water were set equal to the measured reservoir concentrations.

In order to obtain a realistic initial situation at the start of the experimental period, an initialisation run was carried out for the period 1971-2005. For practical reasons, we replicated the years 2001-2005 as meteorological input. This period contains a dry (2003) and a wet (2001) year, and on average the precipitation excess is comparable to the long term average. Fertilizer rates and crop uptake rates for this initialization run were adopted from a previous study by Hendriks et al. (2008).

As previous records were incomplete, we used the sheep data for 2006-2008 in the initialisation runs (1971-2005). It is generally known that peat soils subside due to mineralization. This has consequences for the nutrient budget of the soil profile. Therefore, the nutrient distribution in the soil profile was adapted two times, in 1980 and 1990. For this purpose we used the additional module 'PeatAddit' (Rob Hendriks, Alterra, personal communication). This did not change the height of the peat layer, but we took care of the composition of the organic matter pools.

Table 3.4
Properties of the soil profile at Zegveld (Heinen and Van Kekem, 2011)

Depth (m)	Dry bulk density (kg m^{-3})	C/N ratio
0 – 0.15	562	16.96
0.15 – 0.5	233	15.51
0.5 – 0.75	142	18.19
0.75 – 0.8	142	19.71
0.8 – 0.9	128	19.71
0.9 – 1.8	131	20.22
1.8 – 5.2	131	20.32

Measured and simulated net total N load towards the ditch were of the same order of magnitude, but simulated loads were mostly larger than measured (Figure 3.13). However, the variability in N loads between the three reference and three buffer strip reservoirs was large.

Contrary to sandy regions, the nitrate transport plays only a subordinate role in the total N load at the peat soil site Zegveld. The total load is composed of considerable proportions of ammonium and dissolved organic nitrogen (DON). The behaviour of the ammonium and DON determines the total load to a large extend. This is typical for peat soils where in the sandy areas the majority of the N load consists of nitrate.

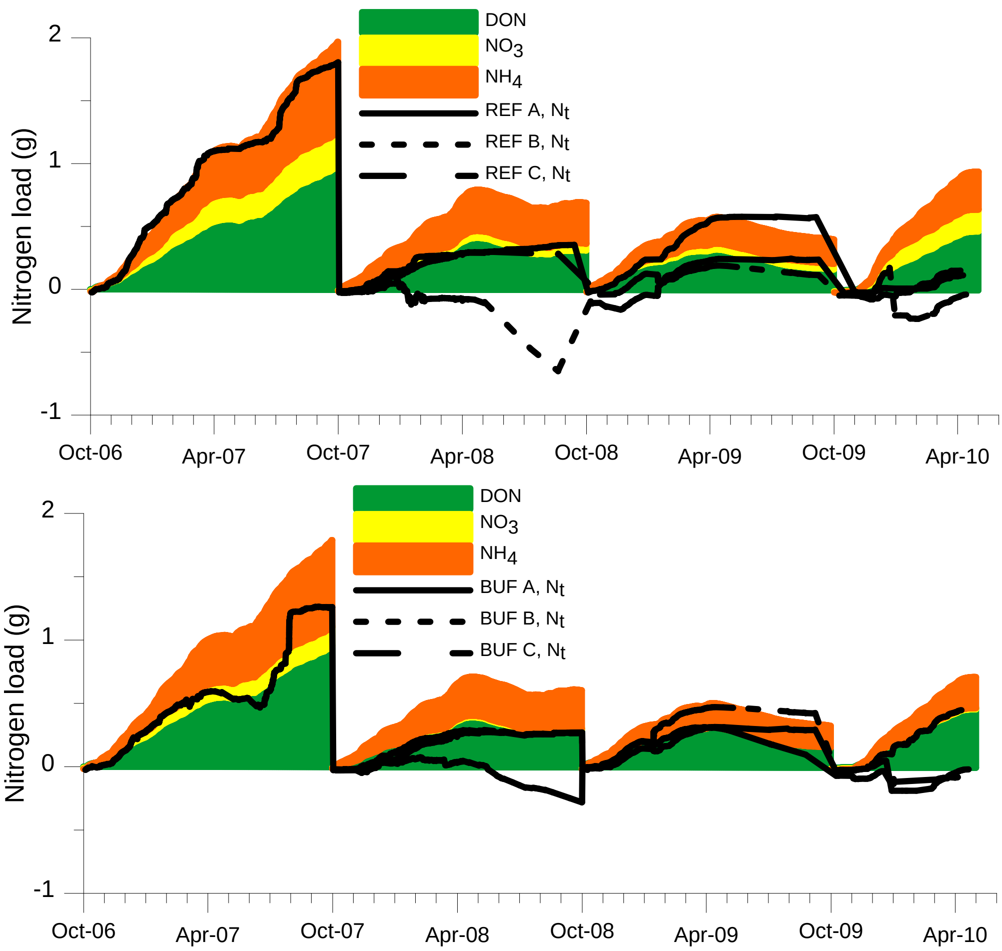


Figure 3.13
Simulated and measured net accumulated N load from field to ditch both for the reference (top) and the buffer strip (bottom) at Zegveld. Simulated total N is split up in nitrate (NO_3), ammonium (NH_4) and dissolved organic N (DON)

Calculated BSE

The calibrated FUSSIM2-ANIMO model was run for 60 years with different BS widths: 2.5, 5, 10, 20 and 30 m, corresponding with a field coverage of 8.3, 16.7, 33.3, 66.7 and 100%. Coverage was calculated as 100% * BS width divided by half of the distance between field ditches (see Figure 3.10). Calibration was conducted for the whole field research period. Extrapolation to long term steady state conditions with newly established equilibria was executed/run by replicating the three representative meteorological years 2006-2008 until a time period of 60 years was reached.

For BS widths up to 10 m a steady-state situation is achieved within the 60 years period (Figure 3.14). The final BSE as a function of BS coverage is given in Figure 3.15. For a BS of 5 m, final BSE equals 17%, corresponding with the BS coverage (see hypothesis in Section 1.3). For BS widths > 10 m, no steady-state was achieved within 60 years. The final BSE for the widths > 10 m was therefore computed in Figure 3.15 with the average loads of the last three years of simulation in Figure 3.14, which is expected to be closest to the final BSE.

Even in case the whole field is treated as a BS, 100% BSE will not be reached due to the background loads of the system. With grazing the maximum BSE would be 50% (Fig 3.15). Except for deposition like in Beltrum, at the peat site Zegveld also mineralisation of peat, upward seepage at the bottom boundary, infiltration of surface water in the summer season and animal excretion during grazing contribute to background loads. So the fertilizer effect of the 5 m BS can now be calculated in the same way as for Beltrum (hypothesis in Section 1.3). The areal fraction of the BS in Zegveld is 5/30 = 16.7%. Based on the ultimate BSE of 50% for a 30 m BS, the expected N load from an non-fertilized strip will be 100 - 50 = 50% of the original load (or of the REF). Hence we can expect a final N load from a 30 m field with a 5 m BS of (5*50% + 25*100%)/30 = 92%. That is a fertilizer effect of 8%, about half the BSE of 17% derived above. Apparently there **is** an additional specific BS effect (see hypothesis in Section 1.3).

Excretion by grazing animals has a great impact on the evolution of BSE (Figure 3.14, interrupted lines): final BSE for a BS of 5 m would reach 22% with grazing instead of 17% without. In case grazing would not have been allowed in the buffer strip, and grassland utilization would have been adapted to this different management regime (only cutting), one would expect higher uptake rates by the crop and thus lower surpluses, which would potentially contribute to an even higher BSE. It should however be noted that the distance between field ditches in this region is very low, ranging from 40 - 60 m, so prohibiting grazing on buffer strips would lead to disproportionate costs due to land loss and necessary fencing.

For P calculated BSE after 60 years for a BS of 5 m was almost 5% (with grazing) and 13% (no grazing). In either case less than BSE for N and less than the fertilizer effect.

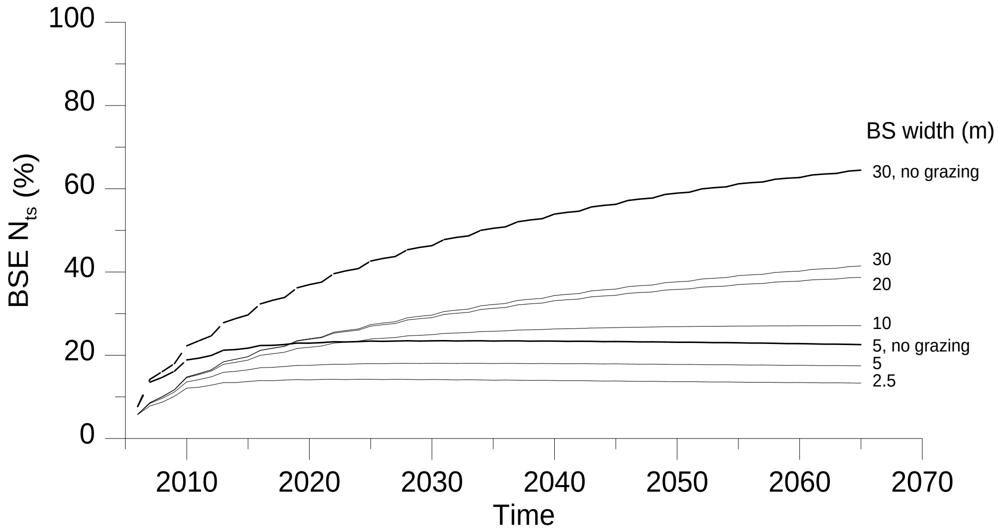


Figure 3.14
Evolution of BSE for N_{ts} as a function of time for five buffer strip widths as obtained with the FUSSIM2-ANIMO model at Zegveld.

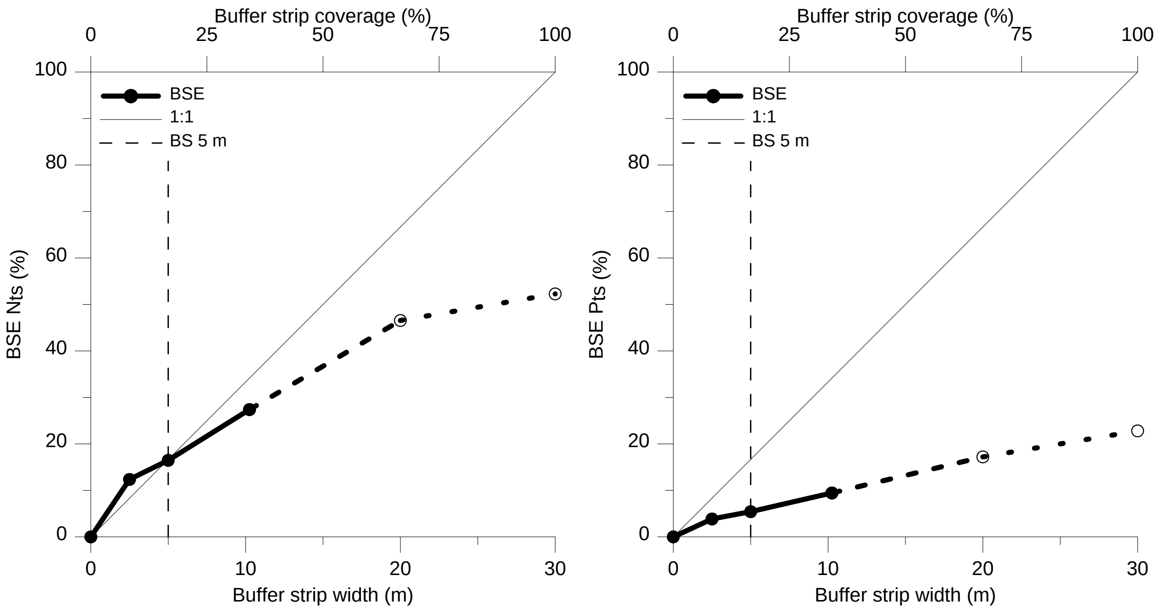


Figure 3.15
Final buffer strip effectiveness for nitrogen (left) and phosphorus (right) at Zegveld, derived from the long term model results of FUSSIM2-ANIMO. Open circles and dotted lines refer to estimates based on the 58-60 year period because the final values for BSE were not yet reached after 60 years according to the model

3.3 Analytical model for extrapolation of BSE values

This section contains a brief description of the analytical model used for interpolation / extrapolation of the BSE to other hydrogeological circumstances and the assessment of the expected range of BSE values.

3.3.1 Description of the analytical model

The effectiveness of a buffer strip is defined as the relative load reduction by a buffer strip compared to a normally managed reference strip according to Section 2.1.6:

$$BSE = 1 - \frac{Load_{\mathrm{BS}}}{Load_{\mathrm{REF}}} \quad [3.1]$$

where $Load_{\mathrm{REF}}$ is the load to surface water for reference conditions and $Load_{\mathrm{BS}}$ is the load with a buffer strip. Note that the water discharge Q is assumed to be equal for REF and BS in the current analysis. Dividing *Load* by Q yields the flow-weighted concentration $\overline{C}$ which converts Eq. [3.1] to Eq. [2.3].

We refer to Appendix 6 for the formal mathematic derivation of the analytical model describing flow, travel time, concentration and loads in a discharge situation. A concise textual description of the model's principles is given below.

The general idea
In Appendix 6 a formal derivation is given of the flow and corresponding changes in concentration for a discharge situation. The general idea is as follows. We consider flow in the saturated zone only. Excess of rainfall passes through the unsaturated zone and reaches the saturated zone. From there it flows along flow paths towards the surface water system (ditch). For certain (simplified) conditions theory allows us to obtain

analytical expressions for the travel time t of the water of each flow path that starts at a certain horizontal distance x from the ditch. So we have an expression for $t(x)$. At the beginning the water has a certain initial concentration C_o. As time progresses decay processes will change the concentration. Assuming a simple first-order decay process the concentration that arrives at the ditch has a concentration equal to

$$C(t) = C_o \exp[-kt] \tag{3.2}$$

If we then replace t in Eq. [3.2] by the expression for the travel time we obtain an expression for C arriving at the ditch as a function of the starting point x of the flow path it had followed

$$C(x) = C_o \exp[-kt(x)] \tag{3.3}$$

Integrating all the concentrations that enter the ditch (multiplied by the discharge Q) then yields the load. This can be done for a situation with and without a BS, so that the BSE can be computed. Note that C_o may be a function of x, e.g., it is zero or low for x in the BS, and high for the REF situation. BSE is then given by

$$BSE = 1 - \frac{\int_0^{x_{BS}} C_{\mathrm{o,BS}}(x)\,\mathrm{d}x + \int_{x_{BS}}^{L/2} C_{\mathrm{o,REF}}(x)\,\mathrm{d}x}{\int_0^{L/2} C_{\mathrm{o,REF}}(x)\,\mathrm{d}x} \tag{3.4}$$

Distinguishing biogeochemical subdomains

The strip adjacent to the water course of fields with a perfect horizontal land surface, has dryer conditions than the remainder of the field. When a field ditch discharges (a part of) the precipitation surplus, the groundwater elevation has an ascending course with distance from the field ditch, but the zone adjacent to a water course (≤ to 3 m) is often wetter than the remainder of the field. When cleaning a field ditch from excess vegetation, the residues and sediments are mostly deposited on this strip which results on the long term higher organic matter contents than the other parts of the field. Both the wetter conditions and the higher organic matter contents may result in higher potential denitrification rates.

Top soils often have higher organic matter contents than subsoils and the organic matter in the top soil is biologically more active than the organic substances in the subsoil. This phenomenon leads also to higher potential denitrification rates in this upper zone.

Potentially four different biogeochemical subdomains can be distinguished with different first order relative decomposition rate constants k (Figure 3.16). The width of the zone adjacent to the ditch is set to X_R and the depth of the top soil with higher reactivity to h.

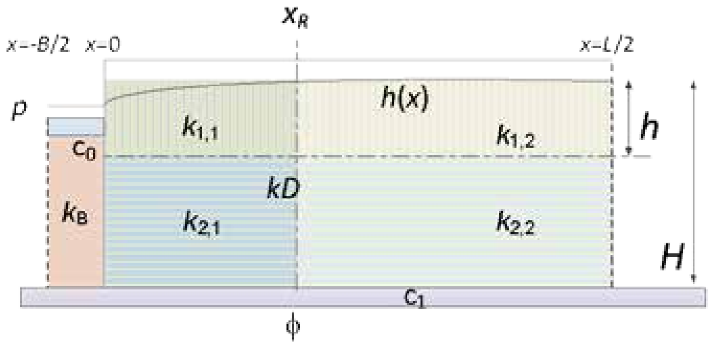


Figure 3.16
Characterization of biogeochemical subdomains with different first order rate constants k (See Box 1 for explanation of the symbols)

The distinction of a subdomain below the ditch is only relevant in cases where the surface water system covers a considerable part of the land surface. In our cases the surface water system covers only a very small part of the surface (< 3/60 = 5% in Zegveld) and we decided to ignore this subdomain in our analysis.

When we distinguish between domains with different reaction rates, the concentration as a function of distance needs account for the travel times in each of the sub-domains. Equation [3.3] then takes the form:

$$
\begin{aligned}
x < X_R \qquad & C(x) = C_o \exp\left[-\left\{k_{1,1}t_{1,1}(x) + k_{2,1}t_{2,1}(x)\right\}\right] \\
x > X_R \qquad & C(x) = C_o \exp\left[-\left\{k_{1,1}t_{1,1}(x) + k_{2,1}t_{2,1}(x) + k_{1,2}t_{1,2}(x) + k_{2,2}t_{2,2}(x)\right\}\right]
\end{aligned}
\qquad [3.5]
$$

where $t_{1,1}(x)$, $t_{2,1}(x)$, $t_{1,2}(x)$ and $t_{2,2}(x)$ are the travel times in the different zones of a particle along a streamline (Figure 3.17).

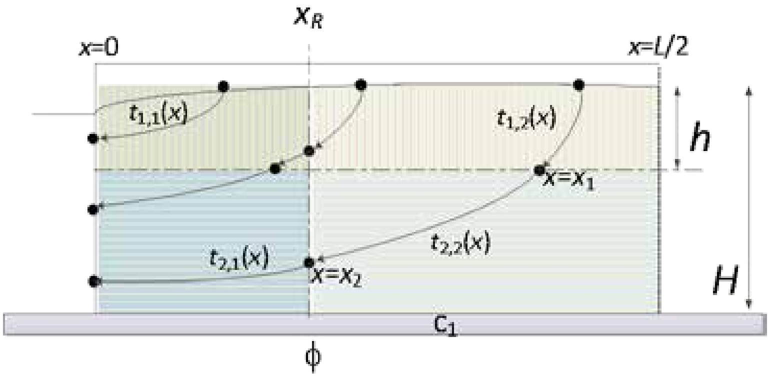


Figure 3.17
Travel times in the biogeochemical subdomains (See Box 1 for explanation of the symbols)

Depending on the position of the starting point of a streamline, the h/H ratio and the $x_R/(L/2)$ ratio, one or more of the $k_{1,1}t_{1,1}(x)$, $k_{2,1}t_{2,1}(x)$, $k_{1,2}t_{1,2}(x)$ or $k_{2,2}t_{2,2}(x)$ terms will cancel in Eq. [3.5]. In general the travel time in a subdomain is calculated as the difference of travel times to the ditch before and after entrance in the subdomain (see $x = x_1$ and $x = x_2$ (Figure 3.17)). The travel time $t_{2,2}(x)$ is calculated as $t(x_1) - t(x_2)$. If a streamline starts at $x = x_0$, the sum of $t_{1,1}(x)$, $t_{2,1}(x)$, $t_{1,2}(x)$ and $t_{2,2}(x)$ equals $t(x_0)$.

Travel time as a function of distance

The analytical model has been applied to four of the five experimental field sites. Due to the presence of pipe drains, the analytical model does not hold for the Lelystad site. The expectations of a low BSE (e.g. Muscutt et al., 1993) for this experimental field site were confirmed by field observations and it was justified to omit model calculations for this location. Each of the field locations has its own specific hydrological circumstances and a specific travel time relation was derived for each of the locations.

Travel time relation for Beltrum

In Beltrum, the phreatic aquifer has a more or less closed bottom. Due to elevation of the regional groundwater system, an influx has be to accounted for. Effective values were derived for both the depth and the length of the discharge domain:

- The regional groundwater occupies some of the groundwater volume at the expense of the volume for local groundwater flow to the field ditch. This has been taken into account by using the effective depth of the discharge domain instead of the depth of the complete aquifer. The effective depth of the profile is proportional to the ratio between discharge and discharge + regional groundwater flow
- A part of the precipitation surplus flows to a greater depth and will form part of the regional groundwater flow (Figure 3.18). The effective field length is calculated from the measured water balance as the field length between ditch and water divide multiplied by the ratio between discharge and net precipitation surplus

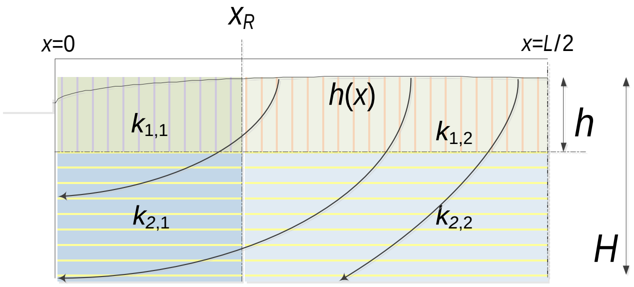


Figure 3.18

Characterization of the biogeochemical subdomains in Beltrum with different first order rate constants k and flow paths, for which travel time relations with distance were derived

The expression for the travel time is sub-divided into different parts to account for the boundaries of the different domains. In general the travel time between the points at $x = x_1$ and $x = x_2$ reads as:

$$t(x_1) - t(x_2) = \frac{\varepsilon H`}{R+S} \ln\left(\frac{L`/2 - x_2}{L`/2 - x_1}\right) \qquad [3.6]$$

The symbols $L`/2$ and $H`$ in Eq. [3.6] have a slightly different meaning form the list of symbols in Box 1. In the Beltrum case $L`/2$ is the field length (m), corrected for the ratio between discharge and total precipitation surplus and $H`$ is the effective thickness of the aquifer (m), corrected for the regional groundwater influx from upstream areas.

Travel time relation for Zegveld

The peat soil in Zegveld is used for dairy farming and has a relatively shallow groundwater level. The dynamic soil matrix leads to dynamic behaviour of the ground level. The difference in the land surface level between summer and winter time can amount up to 9 cm (Beuving and Van den Akker, 1996). This dynamic (seasonal) behaviour can exert influence on the flow pattern and soil biogeochemical processes which are not accounted for in the model. Since we consider only long term steady state conditions, the revealed parameters only relate to averaged conditions A major part of the discharge is conveyed to the field ditch through a relatively thin layer. Contrary to the Beltrum, Winterswijk and Loon op Zand cases, the ammonium and dissolved organic N contribute significantly to the total N load on surface waters. Soil samples revealed no significant differences in potential denitrification rates between the strip adjacent to the ditch and the remainder. Therefore, only a topsoil and a subsoil were distinguished in the transect, and no further distinction between the strip along the water course and the remainder of the field (Figure 3.19).

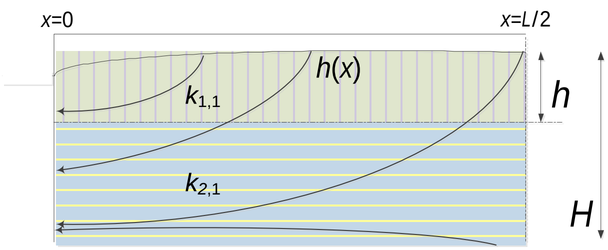


Figure 3.19

Characterization of the biogeochemical subdomains in Zegveld with different first order rate constants k and flow paths, for which travel time relations with distance were derived

The expression for the travel time is sub-divided into different parts to account for the boundaries of the different domains. In general the travel time between the points at $x=x_1$ and $x=x_2$ reads as:

$$t(x_1) - t(x_2) = \frac{\varepsilon H}{R+S}\ln\left(\frac{L/2 - x_2}{L/2 - x_1}\right) \qquad [3.7]$$

Where the meaning of the symbols is explained in Box 1.

Travel time relation for Loon op Zand

For the Loon op Zand case, the general schematization (with four subdomains) as depicted in Figure 3.17 was applied. The aquifer is thin and the field ditch incises the aquitard. For this case the travel time between the points at $x=x_1$ and $x=x_2$ reads as:

$$t_L(x_1) - t_L(x_2) = \frac{\varepsilon H}{Q(0)}\lambda_L \sinh\left(\frac{L/2}{2\lambda_L}\right)\ln\left(\frac{\coth\left(\frac{L/2 - x_2}{2\lambda_L}\right)}{\coth\left(\frac{L/2 - x_1}{2\lambda_L}\right)}\right) \qquad [3.8]$$

Where the discharge rate of water $Q(0)$ which passes the interface at $x=0$ is given by:

$$Q(0) = -\left(R + \frac{\varphi - p}{c_0 + c_1}\right)\frac{\lambda_L^2}{\lambda_B \coth\left(\frac{B}{2\lambda_B}\right) + \lambda_L \coth\left(\frac{L}{2\lambda_L}\right)} \quad [3.9]$$

In which the symbols have the meaning as explained in Box 1. λ_L and λ_B are hydrogeological constants according to:

$$\lambda_L = \sqrt{kD\,c_1} \quad and \quad \lambda_B = \sqrt{kD\frac{c_0 c_1}{c_0 + c_1}} \quad [3.10]$$

Travel time relation for Winterswijk
Winterswijk is characterised by a thin or shallow water saturated layer in a sloping landscape. The field ditch incises tertiary clay formations below the aquifer and the position of the ditch bottom is approximately 0.5 - 1.0 below the top of the tertiary clay formation. Since the sloping aquifer is very thin and the field ditch only discharges water periodically there are no differences in soil moisture conditions as a function of the distance to the water course. Hence, no distinction in biochemical properties of the water saturated layer was made (i.e. one subdomain). The field study showed that the size of the recharge area was influenced by curved iso-lines of groundwater elevation, due to the form of the slope Also the thickness of the aquifer in WW was not uniform but increased with the distance from the field ditch. In the specific case of Winterswijk, both the form of the discharge area (Figure 3.20) and the relation between depth of the aquifer and distance from the ditch had to be accounted for in the analytical model to calculate the travel time as a function of the distance to the water course. For extrapolation purposes we only considered a parallel flow pattern perpendicular to the ditch, and a constant aquifer depth, as one cannot predict the form of field slopes nor transects in advance. However it should be kept in mind that these unpredictable factors influence BSE on specific fields within this hydrogeological class.

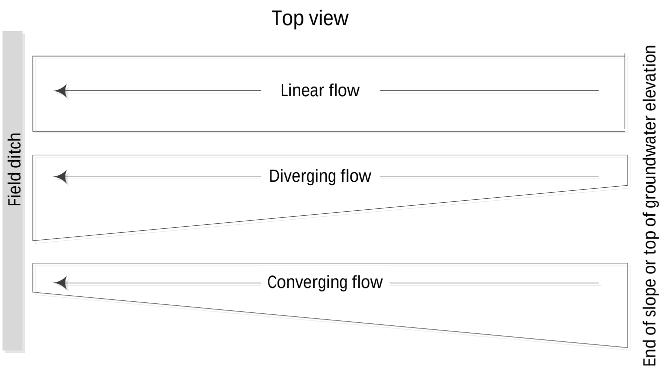


Figure 3.20
Distinction between linear flow, diverging flow and converging flow to describe the Winterswijk groundwater movement to the field ditch in the analytical model

The distinction between travel times of different sections of a streamline is not needed because we do not distinguish between different biogeochemical zones in Winterswijk. It is sufficient to calculate the time before entrance into the surface water of a particle that starts at a distance *x* from the field ditch. The derivation is given in Appendix 7 and the result reads:

$$t(x) = \frac{e}{R}\left\{(H_0 + Lp_2)\ln\left(\frac{L}{L - x}\right) + \left(2B_0\frac{p_2}{p_1} + Lp_2 - H_0\right)\ln\left(\frac{2B_0 + p_1(L + x)}{2B_0 + p_1 L}\right) - 2p_2 x\right\} \quad [3.11]$$

Where the variables are explained in Box 1 and p_1 (=(B_L-B_0)/ L) is the increase of the strip width with distance (-) and p_2 (=(H_L-H_0)/ L) is the increase of the aquifer thickness with distance (-). The strip width and the aquifer thickness at x=L are denoted as B_L and H_L

Sensitivity of BSE
For investigating the parameter sensitivity of the analytical model, elasticity coefficients were calculated for the buffer strip width of 5 m:

$$E = \frac{\Delta BSE / BSE}{\Delta PAR / PAR} \quad [3.12]$$

Where $\Delta BSE/ BSE$ is the relative change of BSE in response to the relative change of a certain parameter $\Delta PAR/ PAR$). The larger the elasticity coefficient is, the more BSE is determined by the parameter. By comparing elasticity coefficients one gets insight in the relative importance of parameters for the final result. Positive E indicates an increase of the parameter value leads to higher BSE values

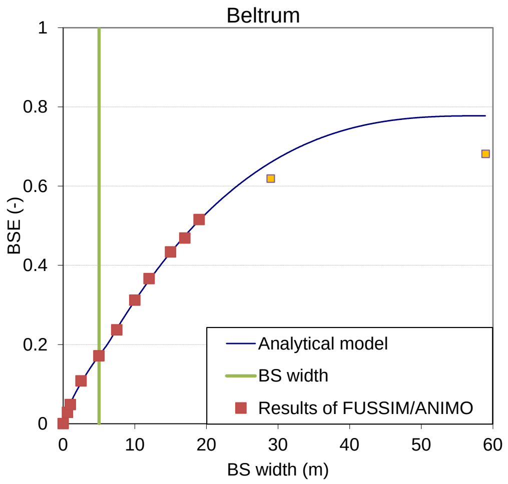
3.3.2 Application of the analytical model to the field sites

The detailed mechanistic simulation model FUSSIM2-ANIMO was applied to the field sites Beltrum and Zegveld (Sections 3.2.4 and 3.2.5) to provide results for calibration of the analytical model. As the analytical model is for steady state situations we used the long term results (60 year) from the dynamic model.

Application to Beltrum
In the calibration of the FUSSIM2-ANIMO model on Beltrum field site data, the influence of the field width has been ignored. Data were lacking to further elaborate this factor in more detail. The simple analytical model enables an analysis of this aspect. The analytical model was applied with the assumption of the transect length of 59 m and the porosity was set at 0.31. The total precipitation surplus (recharge) was estimated at 254 mm yr^{-1}. For establishing the effective depth (Eq. [3.6]) of the profile, the regional groundwater flow in the aquifer was accounted for. The inflow from upstream areas amounted to 31 mm yr^{-1} and the outflow amounted to 136 mm yr^{-1}. The difference in moisture content between start and end of the simulation (see Section 3.2.4) equalled -21 mm yr^{-1} and this amount was added to the downward seepage rate which resulted in a total downward seepage of 106 mm yr^{-1}. The measured average discharge to the field ditch amounted to 146 mm yr^{-1}. The base of the aquifer is situated 20 m below soil surface and 18.7 m below the mean phreatic groundwater level. The water flowing to the field ditch is conveyed in a so-called nested flow system. It should be noted that the streamline pattern of a nested flow system somewhat deviates from the pattern in an aquifer drained by a perfect drain, as was assumed here. In the latter case, the depth of the deepest stream line coincides with the depth of the aquifer but in the former case there may be a difference. On the basis of the water balance of the aquifer, the effective depth of the saturated profile that takes part in the discharge to the field ditch was estimated to be 11.9 m.

The NO_3 input concentration for the regular agriculture part of the transect was set to 30 mg L^{-1} N (Table 2.5; see also Heinen et al., 2012) and the input concentration to the groundwater below the BS was established by calibration. We did not use the data from the field experiment for this because of the relatively short period of the field experiment. The analytical model was derived for long term steady state conditions and new equilibrium between inputs to the soil, organic matter transformation processes and leaching processes.

The first eleven points for BSE as a function of the BS width (Fig. 3.9: x = 0-19 m) were used to calibrate the seven input parameters of the analytical model in Figure 3.17 by using the SOLVER routine of Excel©, including the input concentration to the groundwater below the BS. The remainder of the points in Figure 3.9 relate to greater BS widths, for which steady state was not reached within the simulated time span of 60 years. The result of the calibration is depicted in Figure 3.21, which also presents predicted values for larger strip widths, both for the analytical (line) and the deterministic model (symbols). Note the good correspondence between the calibrated steady state value for C_{BUF} of 6.6 mg L^{-1} and the average measured value in upper groundwater of 8.7 mg L^{-1} (Table 2.5), which is still lagging behind after four years only. The C_{BUF} parameter coincides with the concentration in the upper groundwater to a large extend, but not completely. The relatively short time periods needed for percolation water a BS to reach the upper groundwater are long enough the reduce some of the nitrate leached from the BS. Therefore it is preferable to assess the C_{BUF} value independently and only use groundwater concentrations as estimates for C_{BUF} when no other data are available.



Thickness top layer (h)	1.12 m
Size Reduction zone (x_R)	1.74 m
$k_{1,1}$	0.71 yr^{-1}
$k_{1,2}$	0.22 yr^{-1}
$k_{2,1}$	0.14 yr^{-1}
$k_{2,2}$	0.07 yr^{-1}
C_{BUF}	6.6 mg L^{-1}

Figure 3.21

Calibration of the analytical model for Beltrum on the BSE-values derived by the FUSSIM2-ANIMO model. Orange symbols refer to time extrapolated values which are expected after 60 years

The BSE at a BS width of 5 m results in an effectiveness of 17%. The hypothetical maximum BSE for equilibrium and full BS coverage amounts to 78%. The non-fertilized strip will ultimately reach a new equilibrium in which atmospheric deposition is the only external N source. Leachate concentrations will then be 6.6 mg/L. Sensitivity of the analytical model to the parameters were investigated by calculating the elasticity (Table 3.5).

Table 3.5

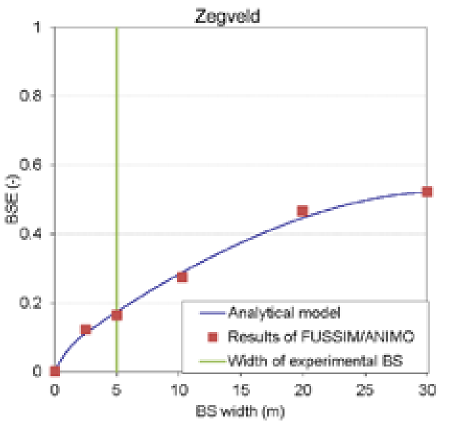
Sensitivity of calculated BSE in Beltrum to parameters of the analytical model (elasticity, see Eq. [3.12])

Parameter	Elasticity	Parameter	Elasticity
Length of field L'	-0.59	**Porosity** ε	0.60
Extend of reduction zone X_R	-0.11	**Rate constant** k **domain (1,1)**	-0.28
Effective thickness of aquifer H'	0.23	**Rate constant** k **domain (1,2)**	0.35
Thickness reactive top layer h	0.37	**Rate constant** k **domain (2,1)**	0.07
Recharge R	-1.02	**Rate constant** k **domain (2,2)**	0.45
Seepage S	0.43	C_{BUF}	-0.29

The recharge rate shows the highest absolute E value, but is negative. An increase of the recharge rate by 1% will results in a BSE decrease of 1.02%. The decomposition rate parameters and the extend of the reduction zone X_R exhibit the lowest elasticity values. Also the elasticity of the background concentration in the percolation water within the buffer strip show a relative low value. The negative elasticity for $k_{1,1}$ is due to the fact that in the reference situation also nitrate is reduced in subdomain (1,1) (see Figure 3.17). The parameters which relate to the hydrological boundary conditions are in Beltrum the most sensitive for BSE assessment.

Application to Zegveld

The peat soil in Zegveld has a high porosity value of 0.92. The distance between the watercourses equals 60 m, so the field length $L/2$ is 30 m. The recharge and net upward seepage rates follow from Section 3.2.5 and amounted to 270 mm yr^{-1} and 7 mm yr^{-1}. A part of the precipitation surplus was discharged by surface runoff and does not take part in groundwater discharge, but. Surface runoff is not included in the model. The total soluble N concentration in the upper part of the groundwater system was estimated based on a series of field observations (Table 2.5) at different distances from the field ditch. The concentration of water leaching to the groundwater outside the BS was estimated to be 11.3 mg L^{-1} (Table 2.5). Most of the total soluble N consists of ammonium and dissolved organic N (see Section 3.2.5). The analytical model in which only the upper layer was distinguished from the subsoil was calibrated on the results of the detailed FUSSIM/ANIMO model (Figure 3.22).



Thickness top layer (h)	0.37 m
$k_{1,1} = k_{1,2}$	0.83 yr^{-1}
$k_{2,1} = k_{2,2}$	0.04 yr^{-1}
c_{BUF}	5.4 mg L^{-1}

Figure 3.22
Calibration of the analytical model for Zegveld on the BSE-values derived by the FUSSIM2-ANIMO model

The C_{BUF} value derived by model calibration is much lower than the measured concentrations in the upper groundwater below the non-fertilized field strip (see Table 2.5), and also lower than the concentration in the reservoir of the BS treatment. The measured concentration in the upper groundwater below the BS can also be influenced by lateral inflow from other parts of the transect. This can be explained by the much larger time period (than 4 years) needed for reaching steady state in the peat soil of the BS, due to the abundance of organic matter. As opposed to mineral soils, the difference in net N withdrawal by the grass between the BS and the REF on the peat soil is small compared with the total amount and yearly turnover of organic N. Hence it takes more time on a peat soil to reach a new equilibrium, than on a mineral soil. As C_{BUF} is the long term concentration of leachate from the non-fertilized strip, it is much lower than groundwater concentration observed after the four experimental years only.

After calibration, BSE calculated with the analytical model is 17.4% for 5 m BS width, which is practically the same as the 17% calculated with the dynamic model.

The thickness of the top layer h is much smaller than the value found for Beltrum. This can be explained the amplitude of the groundwater level which in Zegveld is much smaller than for the Beltrum case. The aerated part of the soil profile, which exhibits a larger microbiological activity than the saturated subsoil, is therefore thinner in Zegveld. Moreover, the difference between the rate constants for the top and subsoil is larger than for the Beltrum case. This difference is explained by the is relatively high potential denitrification rate in the thin top soil because it is governed by the high availability of organic matter in Zegveld. Once the majority of the nitrate has been removed, the denitrification process rate decreases because it is then limited by the nitrate concentration itself.

The sensitivity of the parameters was investigated by calculating the elasticity coefficients (Table 3.6).

Table 3.6
Sensitivity of calculated BSE in Zegveld to parameters of the analytical model (elasticity, see Eq. [3.12])

Parameter	Elasticity	Parameter	Elasticity
Length of field L	-0.69	**Porosity** ε	0.58
Effective thickness of aquifer H	0.15	**Rate constant** k ***top soil***	0.27
Thickness reactive top layer h	0.43	**Rate constant** k ***subsoil***	0.02
Recharge R	-0.56	C_{BUF}	-0.92
Seepage S	-0.01		

None of the factors has an absolute elasticity value greater than one. The background load has the highest influence on the BSE. The rate constant of the subsoil and the upward seepage rate have the lowest influence. The field length, the porosity and the recharge rate all influence the travel time of the groundwater flow. In contrast to Beltrum, the BSE in Zegveld is more sensitive to the factors which govern the residence time in groundwater than the biogeochemical factors. Since the upward seepage rate is very low, an increase of this variable will hardly have any effect on the BSE.

Application to Loon op Zand
The hydrogeological situation at Loon op Zand was described by a thin aquifer on a semi-permeable bottom boundary (aquitard). The distance between the field drains (ditches) was 150 m and the theoretical water divide 75 m from the ditch. However field observations showed the phreatic groundwater level reached its maximum at only 15 m and the observed discharge rates corresponded to a recharge area extending only 10 m from the ditch (Heinen en Van Kekem, 2011; Hoogland et al., 2010). Hydrogeological characteristics were adopted from Hoogland et al. (2010). The phreatic aquifer is relatively thin and due to the dynamics of the groundwater level, the effective thickness varies in time. In our model we however use a constant (effective?) thickness of ca. 1 m. The transmissivity kD was set to 1 m^2 d^{-1} and the recharge rate was estimated to be 0.0008 m d^{-1}. For the thickness of the reactive top layer we could not use the value for Beltrum because this value would exceed the thickness of the aquifer in Loon op Zand. We estimated h to be 0.4 m. The resistance of the semi permeable layer c_1 was set to a value somewhat higher than the value reported by Hoogland et al. (2010), because of the difference between seasonal (Hoogland et al., 2010) and long term (steady state model simulations) recharge rates. In our study we used c_1=625 d. This value results in an ultimate recharge area of 10 m. As can be seen from Eq. [3.9], exact values for the average surface water level (p) and the hydraulic head in the semi-confined aquifer below the aquitard (ϕ) are not needed in our model. Only the difference in head (p - φ) governs the total discharge and the travel times. Based on field observations

(Hoogland et al., 2010) this difference was set to 0.3 m which means that the hydraulic head of the groundwater below the aquitard is 0.3 m deeper than the average surface water level. Since no calibration results of a detailed nitrate leaching model were available for Loon op Zand, the reaction rate constants established for the Beltrum case were applied.

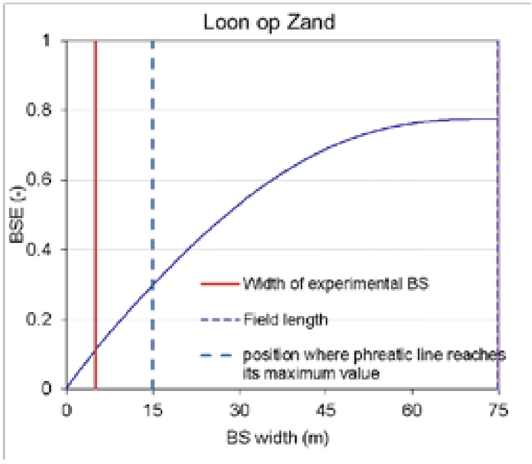


Figure 3.23
BSE-values calculated with the analytical model for Loon op Zand. Parameters for width of reduction zone X_R and decay rate constants ($k_{1,1}$,$k_{2,2}$) were adopted from the Beltrum case

Application of the analytical model yields a BSE value of 11.3% for the strip width of 5 m (Figure 3.23). Alternative strip widths of 2.5 and 10 m would result in BSE values of 5.9% and 20.9%. Elasticity coefficients were established to study the parameter sensitivity (Table 3.7).

Table 3.7
Sensitivity of BSE to analytical model parameters in Loon op Zand (elasticity, see Eq. [3.12])

Parameter	Elasticity	Parameter	Elasticity
Length of field L	-0.16	**Recharge** R	-1.23
Extend of reduction zone X_R	0.08	**Porosity** ε	0.49
Thickness of aquifer H	0.47	**Rate constant domain (1,1)**	0.03
Transmissivity aquifer kD	-0.43	**Rate constant domain (1,2)**	0.06
Resistance aquitard c_1	-1.17	**Rate constant domain (2,1)**	0.02
Surface water level p	3.68	**Rate constant domain (2,2)**	0.38
Hydraulic head deep aquifer φ	-2.93	c_{BUF}	-0.29

From these results it is concluded that apart from recharge, which only determines variability between seasons, the surface water level, resistance of aquitard, and hydraulic head difference between surface water level and deep aquifer are dominant in exerting spatial influence on the BSE results. For the Loon op Zand case, the reaction rates of the soil are less sensitive.

Application to Winterswijk

The hydrogeological conditions at the Winterswijk field are extraordinary for Dutch landscapes, but were also selected for comparison with other European areas. The average field slope is 2% but varies with the exact location. The soil thickness ranges from 40 cm in the strip adjacent to the ditch to 120 cm at the water divide. The long term effective water saturated thickness of the soil layer (H) is estimated to be 30 cm near the field ditch and 60 cm at the water divide (=top of slope). The field study showed that discharge from the BS was much higher than for the REF strip (Table 2.3). This could only be explained by a difference in recharge area. From the isohypse pattern and the water balance of the BS plot it was concluded that the position of the water divide was at 80 m. Discharge from the REF field plot and corresponding recharge area was approximately half the values found for the BS field. This special case requires the adaptation of the expressions for concentration and BSE in Appendix 7. Although this adaptation is useful to investigate the BSE in situations with similar flow pattern to Winterswijk, the linear flow situation should be used for extrapolation, because the pattern of the converging or diverging flow directions is unknown beforehand.

The porosity ε of the sandy topsoil is set at 0.4 m and for long term analyses, the recharge is set at 0.3 m yr^{-1}. Then, the travel time from the outer boundary of a 5 m wide BS amounts to 0.071 yr in case of converging flow and 0.052 yr in case of linear flow. We assumed a first order reaction constant of 1 yr^{-1}, somewhat higher than in the Beltrum case, because in Winterswijk discharge is shallower (the upper top soil is more reactive than the upper groundwater zone at 1.0 – 2.0 depth in Beltrum). The background concentration is assumed to be 20% of the leaching concentrations in the REF field. This assumption is based on the ratio between the nitrate soil moisture concentration of 3.3 mg L^{-1} at 2 m distance of the field ditch in the BS and the average measured nitrate concentration of 17 mg L^{-1} at 21.5 m distance of the field ditch (Table 2.5). The resulting BSE is expressed as a function of the BS width in Figure 3.24. The graph is constructed for two shapes of the recharge area, but for both options we assumed equal hydrological conditions. Since the water layer is rather thin, we have not distinguished separate layers for the flow domain.

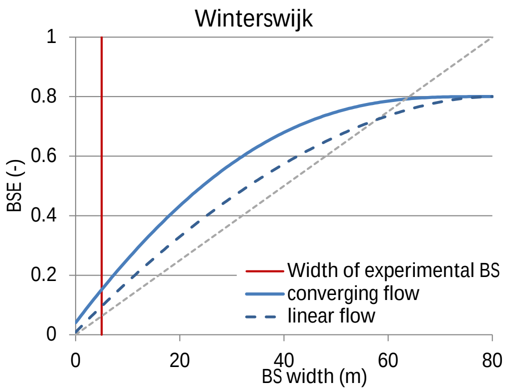


Figure 3.24

BSE-values simulated for the Winterswijk case calculated with two options of the analytical model, converging flow and linear flow

For the 5 m BS, the reduction of the fertilization rate amounts to 5/80 = 6.3%. So the fertilizer effect of the 5 m BS can now be calculated in the same way as for Beltrum (hypothesis in Section 1.3). The areal fraction of the BS in Loon op Zand is 5/75 = 6.7%. Based on the ultimate BSE of 80% for a 75 m BS, the expected N load from an non-fertilized strip will be 100 - 80 = 20% of the original load (or of the REF). Hence we can expect a final N load from a 75 m field with a 5 m BS of (5*20% + 70*100%)/75 = 94.7%. That is a fertilizer effect of 5.3%. The analytical model results yields 15.2% for the case of converging flow and 9.6% for the linear flow option. Apparently there **is** an additional specific BS effect (see hypothesis in Section 1.3). The elasticity provides insight into the relative sensitivity of the BSE to the parameters (Table 3.8).

Table 3.8
Buffer strip effectiveness elasticity of the analytical model parameters for the Winterswijk case (elasticity, see Eq. [3.12])

Parameter	Elasticity	
	Linear flow	Converging flow
Length of field L	0.12	0.95
Thickness of aquifer near field ditch H_0	0.28	0.37
Angle of converging flow p_1		-0.84
Relative increase/decrease of aquifer thickness with distance p_2		0.11
Recharge R	-0.42	-0.48
Porosity ε	0.42	0.48
Rate constant k	0.39	0.46
Background concentration C_{BUF}	-0.25	-0.21

For the linear flow option, the BSE appears to be most sensitive to recharge, porosity and the first order rate constant of nitrate decay. Since the background concentration is low, its influence on the BSE is relatively small. For the converging flow option, the BSE appears to be most sensitive to the field length L and the parameters which define the shape of the converging flow, and less sensitive to the increase of the aquifer thickness with distance and the background concentration.

3.4 Extrapolation of BSE

Van Bakel *et al.* (2007) developed a conceptual framework for the assessment of BSE on the basis of hydrogeological characteristics. Their analysis of hydrogeological factors, combined with surface water characteristics, resulted in the definition of the six hydrogeological classes used in this study (Section 2.1.1). The aerial coverage of each of the classes related to the total area in the Netherlands and related to the Dutch agricultural area is listed in Appendix 8.

A number of processes have been implemented in the analytical model in a conceptual way (section 3.3). The processes are characterised by a maximum of ca. 15 parameters, but for most situations the number of parameters could be reduced (section 3.3). In all cases the field length (or distance between water courses or water divide) is a key factor. The residence time of groundwater is more or less linearly related to the thickness of the aquifer (Van Ommen, 1986).

To gain general insight into the sources of spatial variation of BSE an assessment of the properties which have most influence is needed. For the parameter sensitivity of the analytical model we refer to section 3.3. A next step to such an assessment is an analysis of the variation in BSE values due to the spatial variation of input parameters of the model(s) as can be expected in reality (Table 3.9).

Table 3.9
Measured and simulated BSE for nitrogen load reduction and range of BSE-values based on a parameter variation of ± 20%

		Winterswijk	Beltrum	Loon op Zand	Zegveld	Lelystad
Effectiveness	Field study (3-4 years)	-48.3%	-17.2%±6.4	10.4%	9.8%±6.3	13.9%
	Detailed model (3-4 years)	n.a.	18.5%	n.a.	18-21%	n.a.
	Simple model (long term)	10.1%	17.2%	11.3%	17.4%	
Impact of variability of field characteristics	Field length[1] ± 20%	9% - 10%	15% - 20%	11% - 12%	15% - 20%	n.a.
	Depth of phreatic aquifer ± 20%	9% - 10%	16% - 18%	10% - 12%	17% - 18%	
	Recharge rate ± 20%	9% - 11%	15% - 23%	9% - 16%	16% - 20%	
	Porosity ± 20%	9% - 10%	15% - 19%	10% - 12%	16% - 20%	
	Width of "high reactive" zone along water course ± 20%	10% - 10%	17& - 18%	11% - 11%	n.a.	
	Depth of "high reactive" zone in top soils ± 20%	n.a.	16% - 18%	11% - 11%	16% - 19%	
	Reactivity of soil for reducing nitrate ± 20%	9 – 10%	16% - 19%	10% - 12%	16% - 20%	
	Background load[2] ± 20%	9% - 10%	16% - 18%	11% - 12%	14% - 21%	

[1] given 5 m BS width, so it concerns the ratio BS width / distance from ditch to water divide
[2] variation of background concentration in percolation water

Differences between field observations and the model results are partly explained by:
- Variability of topsoil, subsoil and crop properties within the field
- Field observations result in short term estimates (<4 yr), while model predictions are for the long term
- Abstraction and simplification of the "real world" in a mathematical model

Despite the limitations of the model, the results of the modelling study provide insight into the key factors that determine BSE. As BSE depends on the local circumstances effective BS implementation requires a high level of tailoring to these circumstances. The most important key factors can differ for the sites, as is illustrated in Table 3.9.

The results of the analysis (Table 3.9) indicate that the field length, the seepage and recharge rate and the depth of the aquifer and the "high reactive" zone may have most impact. In order to apply these insights for establishing expected ranges for BSE, we must involve the expected variation of the parameters in reality. We restrict the analysis to those parameters that are either visually perceptible, or quantified in maps or georeferred databases. This excludes the depth of the highly reactive zone.

An analysis was made of the databases of the National Hydrologic Model Instrument (NHI; Berendrecht *et al*, 2008) and of registered surface water characteristics (Massop *et al*, 2006; Van der Gaast *et al*, 2006) for each of the hydrogeological classes for which the BSE was investigated in the field study (Fig. 2.1). (Holland clay, Lelystad was excluded because of the pipe drains.) The surface water database comprises grid-based information on the distance between watercourses derived from basic data by a so-called "moving camera" smoothing technique (Van der Gaast *et al*, 2006). First the fields with drain pipes were omitted from the selection and secondly the distances between water courses were listed. Upward and downward seepage rates are not visually perceptible and information on these variables was derived from the simulation results of NHI. For each of the grids a value of the distance between water courses was extracted as well as the ratio between discharge to water courses and precipitation surplus (recharge). Cumulative frequency distributions were constructed of the distances between the water courses, which at least discharge water during winter. The 10%, 50% and 90% percentile values were selected to establish BSE-ranges (Table 3.10).

Results of model runs with NHI were used to derive cumulative frequency distribution of the seepage rate too. Also for this parameter the 10%, 50% and 90% percentile values are used to obtain an impression of the BSE-values for wider ranges of parameters (Table 3.10). Lower values were rounded off to prevent downward seepage fluxes that exceed the precipitation surplus of the field situations.

The thickness of the aquifer was investigated by Jansen *et al.*, (2012) to obtain depth values for the lower bottom for the calculation units in the STONE model (Wolf *et al.*, 2003) for N and P leaching at the national scale. Although less well underpinned than for the distance between water courses, estimates could be made for median values and upper and lower ranges of the aquifer thickness (Table 3.10).

Table 3.10
Parameter ranges for extrapolation of model results

Parameter	Percentile value	Beltrum	Zegveld	Loon op Zand	Winterswijk
Distance between water courses (m)	Field site	118	60	150	80[a]
	Low 10%	80	35	65	80
	Median 50%	165	50	115	130
	High 90%	400	80	280	230
Upward seepage rate (m yr^{-1})	Field site	-0.106	0.007	-0.25	
	Low 10%	-0.22	-0.26	-0.26	-
	Median 50%	-0.20	-0.03	-0.20	-
	High 90%	0.30	0.20	0.33	-
Aquifer thickness (m)	Field site	11.9	4.8	1	0.4
	Low 10%	7	1.5	3	-
	Median 50%	20	5	7	-
	High 90%	60	13	15	3

[a] field length (distance to top of slope) instead of distance between water courses

In reality the three most important factors determining BSE of Table 3.10 are not independent of each other, but confounded. For instance the lower value of the range of the distance between water courses of the hydrogeological class “deep sand” (Beltrum) most probably corresponds with a lower value of the aquifer thickness. Also, an increase of the aquifer thickness H leads to an increased value of the transmissivity kD at Loon op Zand*.* In practice surface water levels are adjusted seasonally by the water managers or farmers to maintain the downward or upward seepage rates as specified in Table 3.10. The three factors are closely connected. We were not able to execute an in depth study of the complex interdependence between the factors within the framework of this study for BSE-assessment. Instead we made use of the relation between the distance between water courses, the ratio between discharge and precipitation surplus, ditch design criteria, and hydrogeological properties, in order to get a first impression of the expected BSE ranges (Table 3.11). The assessment of BSE ranges should at least include the distance between water courses for three reasons:

- The distance between water courses is the only characteristic that is directly visually perceptible in the field and readily available from georeferred databases, without significant errors.

- The predominant man-made water courses in the Netherlands have been designed according to drainage criteria to enable an optimal agricultural production. Through these drainage criteria distances between water courses are closely connected with hydrogeological characteristics such as the transmissivity of the aquifer and the seepage rate.
- The functioning of a hydrologic system is more sensitive to the distance between water courses, than to the thickness of the aquifer. This proposition can be deduced from the general linearized relationship between the maximum allowed groundwater elevation within an agricultural field Δh_{max} and a critical discharge rate q_{crit} which is used in the design of drainage systems:

$$\Delta h_{max} = q_{crit}\, f(L, H, B, k_{sat}, S)$$

Where k_{sat} is the saturated conductivity in the major flow direction; the other symbols are explained in Box 3.1. Fig. 3.22 gives the relation between the Δh_{max} / q_{crit} ratio, and the distance between water courses for 4 values of the aquifer thickness according to Kirkham (1957). The distance between water courses can be larger at larger values of H to maintain a certain value of the Δh_{max} / q_{crit} value.

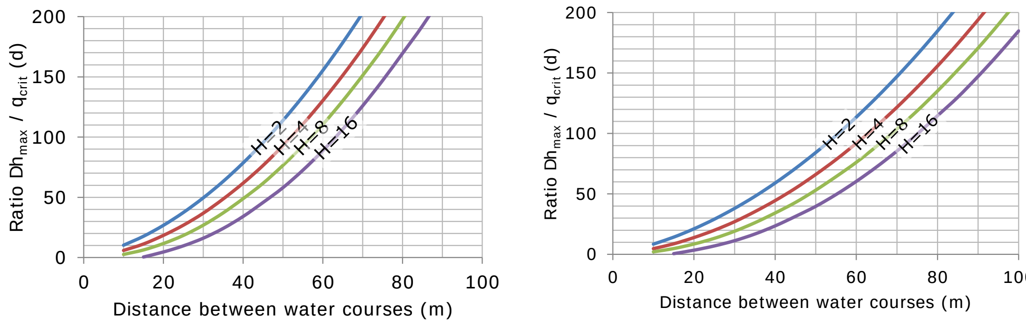


Figure 3.22

Ratio between maximum allowed groundwater elevation Δh_{max} and critical discharge rate q_{crit} as a function of the distance between water courses (wet perimeter 2 m and entrance resistance 1 d) and thickness of the aquifer. Horizontal and vertical conductivity are 2 and 1 m d^{-1} (left) or 3 and 1.5 m d^{-1} (right). Horizontal and vertical conductivities of 1 – 3 m d^{-1} are plausible values for the upper aeolian part of Eastern sand aquifers in the Netherlands (Massop, Pers. Comm.).

Table 3.11

Expected BSE ranges based on the distance between water courses and the analytical model

Distance between water courses (m)	Beltrum deep sand	Zegveld, Holland peat	Loon op Zand, interrupted sand	Winterswijk, shallow sand
Field site value	17.2%	17.4%	11.3%	10.8%
Low 10%	22%	25%	22%	11%
Median 50%	14%	20%	14%	12%
High 90%	7%	14%	9%	13%

The shallow sand profile at Winterswijk is exceptional because of its thin aquifer and slope. The catchment area is slightly sloping and the position of the water divide is more influenced by the land surface elevation and the position of the impermeable layer than by the distance between the water courses. Within this hydrogeological class, distance hardly has any effect on BSE. The highly reactive adjacent strip along the stream will have much greater impact, but such a strip also exists at the REF strip and hence does not contribute to BSE. Noij *et al.* 2012 argue that the residence time in the strips at Winterswijk is too low for retention of nitrate.

The three confounded factors can be partly unravelled by considering the design criteria of the man-made drainage systems in the Netherlands. The design of these systems is based on the maximum frequency of 6 times a year that the groundwater level may exceed a critical depth. This critical depth is 30 cm for grassland and 50 cm for arable crops (Werkgroep Herziening Cultuurtechnisch Vademecum, 1988). The corresponding acceptable critical discharge amounts to ca. 7 mm d^{-1}. The depth of water courses ranges from 80 to 120 cm and the critical height between groundwater elevation and drainage level ranges from 50 to 90 cm bss. Thus, within the design criteria the ratio between Δh_{max} and q_{crit} ranges from 70 to 130 d. The corresponding distance between water courses ranges from 40 to 80 m (Fig. 3.22). These values correspond well with the low percentile values for sandy areas in Table. 3.10. When the distance between water courses is larger than this range, part of the precipitation surplus percolates to deeper groundwater layers and leaves the local groundwater system by downward flow. If the distance between water courses reaches e.g. 160 m, more than 50% of the precipitation surplus leaves the system by downward seepage. When the distance is smaller than the mentioned range upward seepage probably occurs (unless the subsoil is less permeable than the expected 1 – 3 m d^{-1} in Fig 3.22). Inflow by upward seepage increases drain discharge and therefore demands for smaller distances between water courses. This leads to the conclusion that the high distance values in Table 3.10 can only be combined with the low seepage values and the low distance values with the high seepage values. This will obviously limit the potential range of BSE values.

The thickness values in Table 3.10 refer to the maximum thickness of the aquifer if no regional groundwater flow is present. As the infiltration in a region becomes larger, the extent of the regional groundwater flow will increase. Higher regional groundwater flow rates will occupy more volume of the aquifer and leave less volume for the local flow systems. This means that the effective thickness of the aquifer will be smaller for the local system as the downward seepage rate increases. In the Eastern sand areas of the Netherlands, the thickness of the upper aeolian part of the sandy aquifers generally ranges from 3 – 9 m (Massop, Pers. Comm.). After subtracting 1 m for the average depth of the groundwater table, the resulting aquifer thickness for the local flow system ranges from 2 -- 8 m. Based on these data and on the reasoning in the former section, we calculated BSE-values for Beltrum and Zegveld for some typical combinations with the analytical model (Table 3.12).

Table 3.12
Extrapolation of BSE-values based on a wider range of parameters for the hydrogeological class deep sand (Beltrum)

Distance between water courses (m)	Upward seepage rate ($m\ yr^{-1}$)	Thickness local flow system (m)	BSE (%)
80	0	2	12.0
		8	14.2
165	-0.127	2	11.2
		8	12.3
400	-0.203	2	9.3
		8	9.7

Table 3.13

Extrapolation of BSE based on a wider range of analytical model parameters for the hydrogeological class Holland peat (Zegveld), including two values for the concentration in upward seepage water (C_{SEEP}).

Distance between water courses (m)	Upward seepage rate (m yr^{-1})	Tickness local flow system (m)	BSE (%)	
			C_{SEEP} / $C_{0,\ REF}$ = 1	C_{SEEP} / $C_{0,\ REF}$ = 0.5
35	0.152	2	9.7	14
		8	7.1	11.1
50	0.101	2	9.5	12.9
		8	7.0	10.3
80	0	2	14.2	14.2
		8	14.6	14.6

The estimated BSE ranges with interdependence between the model parameters (Table 3.12 and 3.13) are smaller than those in Table 3.11. The results in Table 3.11 therefore need to be considered as maximum ranges. Final BSE for N can vary between 7% and 22% in the sandy areas (deep, shallow and interrupted sand), and between 14% and 25% in the peat area. These values are plausible given the results of the field work (Chapter 2). In specific cases, higher BSE values may be possible, but such specific situations are difficult to predict on the basis of available data on soils, hydrogeology and ditch density.

4 Cost-effectiveness

This chapter is the English translation of the extended summary of the report on cost effectiveness of alternative measures, issued before (Noij et al., 2008).

4.1 Introduction

The goal of this part of the study was to compare the cost effectiveness of BS to the cost effectiveness of alternative measures to reduce nutrient loads to surface water. At the start of this part of the study only the fertiliser effect of BS (Section 1.3) could be included in the model calculations, because the effectiveness of BS (BSE) had yet to be determined. The costs, however, could already be determined (€ per hectare per year). This enabled us to calculate how effective BS would have to be in comparison to the alternative measures. Effectiveness was expressed in absolute load reduction, kg N or P per hectare per year and cost effectiveness in € per kg N or P.

4.2 Selection of alternative measures

The selection of most promising alternative measures was based on Van Os et al. (2009), a survey of characteristics of measures collected in a database, based on literature data, expert judgement and previous research. A distinction was made between a series of source measures, hydrological measures, and a constructed wetland or wetland buffer strip. Within the source measures, we further distinguished between a number of different dairy farming and arable farming systems. For dairy farms the measures reduced grazing or zero grazing and P-mining were compared with BS. For arable farms, in addition to two P-mining[5] variants (on wheat versus in rotation), the effects of stricter nutrient application standards, lower fertiliser rate and spring application of animal manure on clay soils were calculated. The hydrological measures were blocking surface runoff, removing pipe drainage, conventional pipe drainage and controlled deep pipe drainage.

4.3 Methodology

The effectiveness of alternative measures was estimated using model calculations and literature survey. Costs and loads after application of the measure were compared with the costs and loads in the original situation. The calculated effectiveness of measures was less at farm level than at field level, because the calculated measures were only applied to part of the farm acreage. Buffer strips covered 5-20% of the acreage, P-mining was applied to 15-30% and reduced or zero grazing to the full grass acreage (sand 70%, clay 90% or peat 94%).

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[5] With P-mining or phytoextraction one strives for maximum net P withdrawal by the crop with normal fertilizer rates, except for P (or animal manure). See Van der Salm et al., 2009.

No integrated model instrument for the farm level was available. The national model for evaluation of manure legislation was not suitable for farm level (STONE, Wolff et al., 2003). The costs of source measures were determined by means of calculation with the farm models BBPR (dairy; Schils et al., 2007) and MEBOT (arable; Schreuder et al., 2008), and the costs of hydrological measures, constructed wetlands and wetland BS were estimated by experts. The fertiliser rates as calculated with the same farm models were used as input for the leaching model ANIMO (Groenendijk et al., 2005) to calculate the surface water loads for the source measures. For each combination of farm type and measure different combinations of soil, hydrology and crop (plots) were studied, but always one plot at a time. This allowed us to evaluate the effect of soil and hydrology (spatial variation) on the effectiveness of measures. The fertiliser effect of BS was calculated by area weighted averaging of the load reduction by non-fertilised grassland (in the BS) and normally fertilised grassland (adjacent to the BS). For the load reduction from grass buffers adjacent to maize and arable farming plots, the load reduction of the most similar non-fertilised grass plot was taken.

As a result of conceptual differences and differences in model parameters, differences in crop withdrawal and surplus between the farm models and ANIMO were unavoidable. These differences were analysed and largely explained, so the impact on the leaching could be indicated. This proved to be a time-consuming procedure.

SWAP is the hydrological model used, which generates input for the leaching model ANIMO. The hydrological measures are therefore first calculated with SWAP, and then the hydrological output is used as input for ANIMO in order to calculate the effects on the N and P loads. The effects of constructed wetlands and wetland BS were estimated based on literature survey and the expected input load levels in the five regions in which the field research was carried out.

As this model study was conducted without empirical validation, the results must be considered preliminary. This means that the results may be used to indicate the performance of measures and set priorities in the selection of measures for pilot projects and research. The untested model results are, however, insufficient to substantiate policy measures at this stage, other than to stimulate innovations and trials. Anything more would require more farm situations and spatial variation in the approach, in addition to validation with empirical data.

If the government chooses to follow the approach of application of tested models to find optimum mitigation packages for different soil, water and farming situations, then we recommend developing a substantively consistent and efficient model toolkit to be derived from the models used here. The advantage of such an instrument, would be that studies can be better geared towards specific regional situations or catchment areas or sub-areas. The remote linkage between models used here proved to be extremely elaborate and time-consuming, with all the risks of human error that this implies.

4.4 Effect of soil and hydrology

Surface water loads are usually more strongly determined by the properties of the soil and hydrology (plot) than by the effects of measures. The result is that the most cost effective measure for any given location is always unique.

In the starting situation, N and P loads[6] varied between the plots from approximately 10 - 50 kg N per hectare per year and 0.5 - 5 kg P per hectare per year (with one extreme exception of 17 kg P per hectare per year). Plot differences are caused primarily by differences in background load (upward seepage and mineralisation). Therefore, load reductions are expressed not in relative (%) but in absolute terms (kg per hectare per year).

4.5 Source measures

Source measures have little if any effect on P load to surface water, because this load is primarily determined by the P status of the soil and by the hydrology. Only with the source measure P-mining can a significant reduction in the P load (double-digits in %) be achieved in the long run (order fifteen years). The largest P-load reduction is achieved with P-mining on wheat fields (P-load reduction of 1 kg per hectare of P-mining fields per year = 0.3 kg per farm hectare per year), but this proves to be extremely expensive. The reduction through P-mining increases with the original P status (1), the accumulated net P withdrawal, i.e. the mining period (2), and the proportion of shallow transport routes (3). Our calculations apply to a mining period of fifteen years (2), and we selected a representative sample of Dutch sandy soils (1 and 3). This means we did not specifically calculate the effect for P-leaking soils.

For P-leaking soils, the P-load in the starting situation will be higher, and accordingly so will be the effect of P-mining. If the P-fertiliser rate on the rest of the farm increases due to P-mining, as with intensive dairy farming on sandy soil, then the P-load reduction (0.25 kg per hectare mined grass per year) will be partly compensated by the increase in leaching on the other plots.

The ideal starting situation is one in which the difference in P status between the P-mining plot and the rest of the farm is the greatest, and the areal percentage of P-leaking soils where P-mining is to be applied is the least. P-mining also has a beneficial effect on the N load from the fields in question, and in most cases this beneficial effect for N also holds for the farm level.

The negligible effects of source measures on the N loads are due primarily to the amount of fertiliser applied and the amount of the N surplus. The N load to the surface water is reduced as less fertiliser is used and as the N surplus drops. In arable farming, spring (instead of autumn) application of fertiliser on clay soils makes the biggest contribution to N reduction (3 kg per hectare per year, or nearly 20%), followed by 20% lower fertiliser rate (2 kg per hectare per year), and stop slurry application (1 kg per hectare per year). Lowering the application standard has even less of an effect, but this is also due to the fact that the standard is not fully applied in the starting situation.

In dairy farming, N-load reduction of reduced grazing was underestimated because N-withdrawal from grass cuttings was too low in the leaching calculations. We only calculated a substantial drop in the N-loads with zero grazing on sandy soil (N load reduction of 3 kg per hectare grass per year = 2 kg N per farm hectare per year). We estimate the actual load reduction at the farm level of zero grazing to be 2 (clay and peat) to 3.5 kg (sandy soil) N per hectare per year.

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[6] If the entire precipitation surplus of 300 mm per year would be discharged to the surface water, the norm without retention corresponds to an N-load of 6.6 kg ha^{-1} j^{-1} and a P-load of 0.45 kg ha^{-1} j^{-1}

4.6 Buffer strips

The fertiliser effect of BS on the calculated P-load was virtually nil. This should not be surprising, given that the P-leaching from the buffer strip is determined by the soil P status, which only changes in the long term. In the long term, the soil in the buffer strip will be mined out, at which point a fertiliser effect on the P-load can be expected.

The fertiliser effect of 5% BS on the N-load on arable farms is in the order of 0.5 kg N per hectare per year, approximately 2% of the total N-load in the starting situation. This low percentage is the consequence of a high background load.

The fertiliser effect of a 5% (grazed) buffer strip on the N load on dairy farms was small (<0.25 kg N per hectare per year, <2%). The fertiliser effect of a zero grazed grass buffer strip on sandy soil was over twice as high as with a grazed buffer. On the farm level, an amount of slurry transported away from the farm, corresponding with the application standard for the buffer strip area, resulted in less reduction than zero grazing of BS.

4.7 Hydrological measures

The effectiveness of hydrological measures was only calculated at the field level. Scaling up to the farm level would be pointless because the effectiveness and costs of the measure per hectare would not change. The hydrological measures are generally much more effective than the source measures, but the effectiveness is strongly dependent on the starting situation, and generally differs for N and P.

Blocking shallow trench and surface runoff is an effective measure on sandy soil without tile drains, with a load reduction in the order of 1 kg P per hectare per year (40%) and 2 kg N per hectare per year (10%). On pipe-drained sandy soils, the P-load reduction was much less, just over 10%. Blocking surface runoff is not possible on peat soils and not effective on clay soil.

Conventional pipe drainage works for P on all mineral soils, and in some cases also on peat soil. Load reductions are, on sand, 1.3 kg P per hectare per year (>60%) and on clay 0.6 kg P per hectare per year (>50%). On sandy soil, however, this measure leads to a dramatic increase in the N-load (10 kg N per hectare per year, 150-250%). On clay soil, drainage also decreases the N-load (6 kg N per hectare per year, 30-40%). Clay soils, however, are largely already tile drained.
For peat, calculations were carried out for only two plots, and the effects of conventional pipe drainage appear to be variable, an increase of the loads on the grass plot with downward seepage, and a small reduction of N-load and substantial reduction of P-load on the relatively uncommon arable peat plot with maize.

Controlled deep drainage works better than conventional drainage on sandy soil, because the N-load increases much less (in some cases even decreasing). Averaged over the year, level control does not lower the groundwater level, and the drain flow path is longer than with conventional drainage. In the model calculations, P-load decreased somewhat less than with conventional drainage (average 1 kg P per hectare per year, as compared to 1.3 kg with conventional drainage) because the calculations were based on a fixed level of 60 cm below ground level, which in some wetter periods leads to crop damage and extra P load. In practice, this is easy to remedy, because there is more room to further refine the level management of the controlled system with a more beneficial effect on crop production and P-load. On clay soil, this system is less effective than conventional drainage because of the fixed level.

Stopping drainage results in a significant N-load reduction, in some cases by over 40 kg per hectare per year, but the P-load increases significantly, in some cases by more than 1 kg per hectare per year.

4.8 Purification systems

Constructed wetlands, and to a lesser degree wetland BS, appear to be effective according to the calculations. Reduction of the P-load is 0.3-3.9 kg P per hectare of farmland per year (15-80%), and N-load reduction is 2.2-19.7 kg N per hectare per year (20-50%). P-reduction is greatest when the P-load from agriculture is high and if the purification system is intensively managed (yearly mowing and restoring every six years). The P-load of the purification system depends in part on the selected areal ratio of purification system to farmland. In our calculations, this ratio is on the high side (gross 5% and net 3.5%), and is therefore best suited to relatively high input load areas (4 kg P per hectare farmland per year).

A condition for calculated N load reductions is that there must be sufficient biodegradable carbon in the system. In an unmanaged system, that will usually be the case, but with intensive management for maximum P load reduction, this could become a limiting factor. This means that in the design of a purification system, the limiting nutrient in the water system in question, as well as the load in the starting situation, must be taken into account. In addition, the N and P status of the soil in which the system is to be constructed plays a role.

4.9 Costs

The costs of the source measures were calculated for only a small number of model farms, three dairy farms and four arable farms (Table 4.1). The consequence of this approach is that we cannot give any indication of the degree of uncertainty, while in practice, there is of course an enormous variation in farm management and strategy. Some measures may be more or less effective depending on these variables. This means tailoring measures should also involve the costs for a particular farm. The introduction of reduced grazing on a dairy farm is, for example, not cost-effective as a measure purely for surface water quality, but if the dairy farmer is planning to introduce this anyway for other reasons, the load reduction is an added bonus. The same can be said for instance for removal of crop residues in arable farming. If the costs of the measure are assigned to surface water management, it may be much too expensive, but if the farmer can sell the biomass for energy production, the situation could be different. Similar arguments may be set up for other measures.

The calculated costs of BS for the less intensive dairy farms on clay and peat soils were very low, because they are self-sufficient for roughage, in contrast to more intensive dairy farms on sandy soils. The minor increase in the purchase of roughage was compensated for by lower purchase of fertilisers (clay) or feed concentrates (peat) and minor changes in the use of contract labour.

For arable farms, the costs of source measures with minor loads reduction, are € 0-100 per hectare per year. For dairy farms, the source measures are more expensive. Only the costs of P-mining on a limited area come anywhere near the costs of other measures. However, if the P-mining area exceeds the threshold, where grazing needs to be reduced, the costs of P-mining skyrocket.

Table 4.1

Costs of measures to reduce nutrient loads from agricultural lowlands to surface waters

Sector	Rank No.	Rank € ha^{-1} j^{-1}	SANDY SOILS	CLAY & PEAT SOILS[1]
Source measures Arable farms[1]	1	0-50	Application standard-20% P-mining in rotation BS	Spring application Application standard-20% P-surplus 0
	2	50-100	P-surplus 0 No fertiliser	No fertiliser BS
Source measures Dairy farms[1]	1	0-50	BS	BS
	2	50-100		
	3	100-200	P-mining grass/maize	
	4	200-400	Reduced grazing	Reduced grazing
	5	>400	P-mining with zero grazing	
Hydrological measures & Purification systems	1	0-50	Stopping drainage (yield loss not accounted for)	Stopping drainage (yield loss not accounted for)
	2	50-100	Blocking surface runoff	Blocking surface runoff
	3	100-200	Conventional drainage, Wetland BS minimal management	Conventional drainage, Wetland BS minimal management
	4	200-400	Controlled deep drainage, Constructed wetlands, Wetland BS with mowing in September	Controlled deep drainage, Constructed wetlands, Wetland BS with mowing in September
	5	>400	Wetland BS restored at 6-year intervals	Wetland BS restored at 6-year intervals

[1] Arable farming only on clay soils, dairy farming both on clay and peat soils

The hydrological measures blocking surface runoff and conventional pipe drainage are well affordable; the agronomic advantages of conventional drainage were not accounted for. Controlled drainage is not common in the Netherlands, so there is little experience with its maintenance. Therefore our experts estimated relatively high costs, that have not yet been confirmed in practice. The bottom line is that further research must demonstrate the conditions under which the system does and does not continue to perform well in the long term.

Purification systems may be expensive, but they are also effective (see next section).

4.10 Cost-effectiveness

The cost effectiveness in € per kg N or P was calculated as the quotient of costs in € per hectare per year of the measure and the load reduction in kg N or P per hectare per year by application of the measure. The variation in cost effectiveness was the result of the variation in effectiveness between the calculated plots and the variation in costs between the model farms. Tailoring measures to specific farm situations would reduce costs and increase effectiveness. The ranking in Table 4.2 is based on the cost effectiveness, averaged over plots. The cost effectiveness of the measures differs between N and P, because the effectiveness also differs. In order to be able judge the ranking of the cost effectiveness of the measures in Table 4.2 we made a comparison with the cost effectiveness of nutrient load reduction by extra measures in wastewater treatment plants, varying between 32 and 134 € per kg P-equivalent (1 kg P-equivalent is 1 kg P or 10 kg N; Van Soesbergen, 2007). The rankings 1 and 2 are therefore considered cost-effective.

The calculated cost effectiveness was limited to the benefit for surface water quality. The benefits for other environmental themes or societal goals, such as groundwater quality, reduction of pesticide loads, reduction of greenhouse gas emissions, increasing biodiversity and ecological connectivity, attractive landscape, stabilising ditch banks, etc., were not accounted for. Measures that did not prove to be cost effective in this approach might therefore still be meaningful measures. If measures are adopted for other reasons, they can still make a (small) contribution to the quality of the surface water.

For dairy farms, no cost effective source measures were found. Although the N load reduction from reduced grazing was underestimated in the model calculations, correcting for this had very little impact on this conclusion. Reduced grazing of livestock as a specific measure for the quality of surface water is an extremely expensive measure. P-mining, on the other hand, could be further investigated to find conditions resulting in a higher cost effectiveness. More favourable cost effectiveness is expected on specific farms where P-mining could be focused on a small area with strongly P-saturated soil, and where the remainder of the farm fields have limited soil P status.

Despite the limited effectiveness of source measures, some of them still proved to be cost effective for arable farming because their costs were also limited. In arable farming, postponing slurry application to spring on clay soils (spring application) is the most interesting source measure, because it results in a considerable reduction of N-load. Likewise, reducing the application standard is not particularly expensive and produces noticeable N-load reduction. In addition, the P-surplus can be reduced at no cost by limiting the use of artificial fertiliser, though this will not reduce P-leaching. The only cost-effective measure that reduces P-load is P-mining in rotation. Though P-mining with grain is more effective, it is also much more expensive.

Although hydrological measures or purification systems proved to be more cost effective for surface water than source measures, they can never serve as a replacement for generic source policy. Generic source policy prevents the soil to become the future source of nutrient loads. Furthermore, the source policy serves more objectives than just surface water. It is more accurate to state source measures are insufficient to protect the surface water, and supplemental measures are necessary.

Purification systems such as Constructed wetlands and wetland BS can be cost effective for reaching good water quality. In addition, these systems offer the potential to join up with other goals such as water storage, biodiversity and landscape. Of course, the cost effectiveness calculated depends on a series of selected assumptions. Significant assumptions include the way in which the costs of the land are calculated, the selected management and maintenance, the input load level of the purification systems, the corresponding ratio between farmland catchment area and purification system area, and the priority-setting between N and P. The most important conclusion is that these systems offer potential, and that the best design in terms of size, load and maintenance will have to be determined on a case-by-case basis. In addition, further policy and practical initiatives will have to be developed for this option, because they demand an approach that goes beyond the farm scale. How do we set up the fertiliser policy for a group of farmers that selects a purification system? How does that group of farmers apportion the costs and benefits between the farms affected by installation of such a system and farms that are not?

Table 4.2
Cost-effectiveness of alternative measures. Measures in rank 1 and 2 are considered cost effective by comparison with extra measures in wastewater treatment plants, see text for further explanation

Sector	Rank	€ / kg N	€ / kg P	SANDY SOILS		CLAY AND PEAT SOILS[1]	
				N	P	N	P
Source measures Arable farms	1	<10	<100	Application standard -20%	P-mining in rotation	Spring application , P surplus 0	No reduction
	2	10-20	100-200			Application standard -20%	
	3	20-50	200-500	P-surplus 0	P-surplus 0	No fertiliser	No reduction
	4	50-200	500-2000	P-surplus 0 P-mining	P-surplus 0 P-mining grain		No reduction
Source measures Dairy farms	4	50-200	500-2000	Reduced grazing P-mining grass	P-mining maize	No reduction	No reduction
	5	>200	>2000	P-mining & SF P-mining maize	P-mining maize P-mining grass	No reduction	Reduced grazing
Hydrological measures	1	<10	<100	Stopping drainage			Pipe drainage (peat)
	2	10-20	100-200	Controlled deep drainage	Blocking surface runoff Controlled deep drainage		
	3	20-50	200-500	Blocking surface runoff Controlled deep drainage		Blocking surface runoff & Controlled deep drainage (clay). Pipe drains (clay/peat)	
	4	50-200	500-2000	Controlled deep drainage		Blocking surface runoff & Controlled deep drainage (clay)	
	5	>200	>2000			Blocking surface lev. (clay)	
Purification systems				LOW	LOAD	HIGH	LOAD
	2	10-20	100-200			Constructed wetlands	Constructed wetland, September mowing or restoration at six year intervals Wetland BS, restoration at six year intervals
	3	20-50	200-500	Constructed wetlands	Constructed wetlands, Wetland BS, September mowing or restoration at 6 year intervals	Wetland BS	Constructed wetlands, unmanaged Wetland BS, September mowing
	4	50-200	500-2000	Wetland BS			

[1] Arable farming only on clay soils, dairy farming both on clay and peat soils

The hydrological measures also offer considerable potential for cost-effective reduction of nutrient loads to surface water, but here again, they demand a site specific approach. Hydrological measures can even have undesired effects, such as increasing N-load in the case of conventional drainage on sandy soils. In brief:

- Blocking surface runoff primarily contributes to reducing P-loads on sandy soils. It is cheap and feasible for individual farmers to carry out independently, but may be counter to the farmer's intuition that the water has to get off the land as fast as possible.
- Conventional drainage primarily reduces P-loads on sandy soils, but increases N-loads substantially. On clay soils, this measure also reduces N-loads, but clay soils are generally already drained. Stopping drainage has the opposite effect.
- Controlled deep drainage is mainly interesting as a measure for sandy soils, because in contrast with conventional drainage, along with the P-load this system also reduces the N-load to the surface water. The system could come out more beneficial in practice, than in the model calculations, because 1) the level can be adjusted flexibly to weather and other conditions, 2) the agricultural benefits of improved drainage and moisture supply were not included in the calculations, and 3) maintenance costs may have been overestimated due to little experience with the technical functioning of this system.

When the study on cost effectiveness of alternative measures was executed, only the costs and the fertiliser effect of BS could be calculated. The fertiliser effect is the result of the (area weighted) reduced fertiliser rate on the plot due to the introduction of a BS (Section 1.3). On sandy soils, the cost effectiveness of the fertiliser effect of BS comes out in the 4th rank of Table 4.2 (50-200 € per kg N or 500-2000 € per kg P per year). This means there must to be a considerable specific BS effect (Section 1.3) in addition to the calculated fertiliser effect of BS, before BS can 'compete' with the alternative measures in ranks 1-2. Load reductions by BS would have to be on the order of kilogrammes per ha per year for N, and tenths of kilogrammes per ha per year for P, which in both cases amounts to multiples of 5% of the original load. Table 4.3 shows the required total BSE for BS to compete with the alternatives of ranks 1-2.

For arable farming on clay soils, a significant effectiveness (BSE) greater than 20% of original load is required; in other situations, a 10% BSE would appear to be sufficient. This is still greater than the calculated fertiliser effect of BS (< 2% with 5% soil fraction). For clay soils, no specific BS effect is expected, because these soils generally have pipe drainage.

For dairy farming on clay and peat, the calculated costs of grazed BS are so low that their cost effectiveness is comparable to the alternatives even with low load reduction (0.1 kg P and 1 kg N ha^{-1} yr^{-1}, approximately 5-10% (BSE) of original load reduction). For a substantial reduction of the load, the BSE would have to be much higher than the fertiliser effect.

Table 4.3

Required N- or P-load reduction by BS (5% areal fraction on sand and clay soil, 10% on peat soil) for a cost effectiveness that can compete with the alternative measures (ranks 1 or 2 in Table 4.2)

Situation		BS costs	Original N-load	Cost effectiveness ranking; € per kg N		Original P-load	Cost effectiveness ranking; € per kg P	
		€ ha^{-1} y^{-1}	kg ha^{-1} y^{-1}	1; <10	2; 10 – 20	kg ha^{-1} y^{-1}	1; <100	2; 100 – 200
				Required N-load reduction kg ha^{-1} y^{-1}			Required P-load reduction kg ha^{-1} y^{-1}	
Arable farms								
	CZK[1]	135	19.5	> 13.5	6.3 – 13.5	0.69	> 1.35	0.68 – 1.35
	ZWK[2]	41	15.7	> 4.1	2.1 – 4.1	0.32	> 0.41	0.21 – 0.41
	ZON[3]	40	28.4	> 4.0	2.0 – 4.0	4.22	> 0.40	0.20 – 0.40
	NON[4]	22	29.1	> 2.2	1.1 – 2.2	1.64	> 0.22	0.11 – 0.22
Dairy farms								
	Clay	0	15.7	0	0	7.90	0	0
	Sand	12	19.1	> 1.2	0.6 – 1.2	1.42	> 0.12	0.06 – 0.12
	Peat	0	6.1	0	0	1.30	0	0

[1]Central marine clay area; [2]Southwest clay area; [3]Southeast sand area; [4] Northeast sand area

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Appendix 1 Additional site information

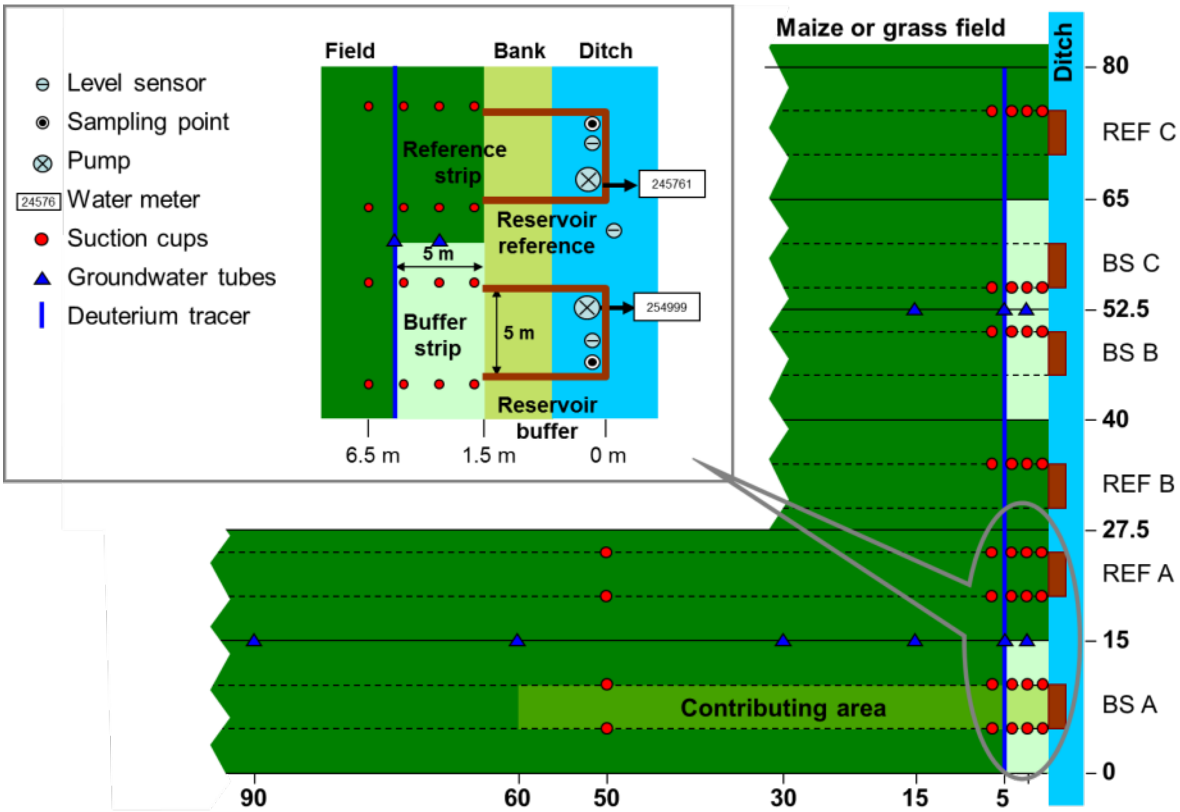


Figure A1.1
Lay-out (in-set is not at scale) of the experimental site at Beltrum

Table A1.1
Rainfall (mm) and precipitation excess (PS, mm: rainfall minus Makkink reference evapotranspiration) at the five experimental sites for the leaching periods (Oct - Apr) and intermediate summer periods

Period	Zegveld		Beltrum		Winterswijk		Loon op Zand		Lelystad	
	Rain	PS	Rain	PS	Rain	PS	Rain	PS	Rain	PS
Oct 06 – Apr 07	585	476	489	382	481	374	442	328	531	419
Apr 07 – Oct 07	648	177	513	47	564	98	403	-74	475	-2
Oct 07 – Apr 08	363	257	391	285	448	342	386	276	329	228
Apr 08 – Oct 08	404	-68	358	-123	219	-261	473	5	339	-146
Oct 08 – Apr 09	348	244	273	176	330	233	318	209	346	248
Apr 09 – Oct 09	282	-228	263	-252	336	-179	n.d.	n.d.	n.d.	n.d.
Oct 09 – Apr 10	604	505	363	265	469	370	n.d.	n.d.	n.d.	n.d.

n.d.: not determined

Appendix 2 Deuterium breakthrough curves

In the main text the deuterium breakthrough curves for Beltrum were given. Below follow those for the other four sites.

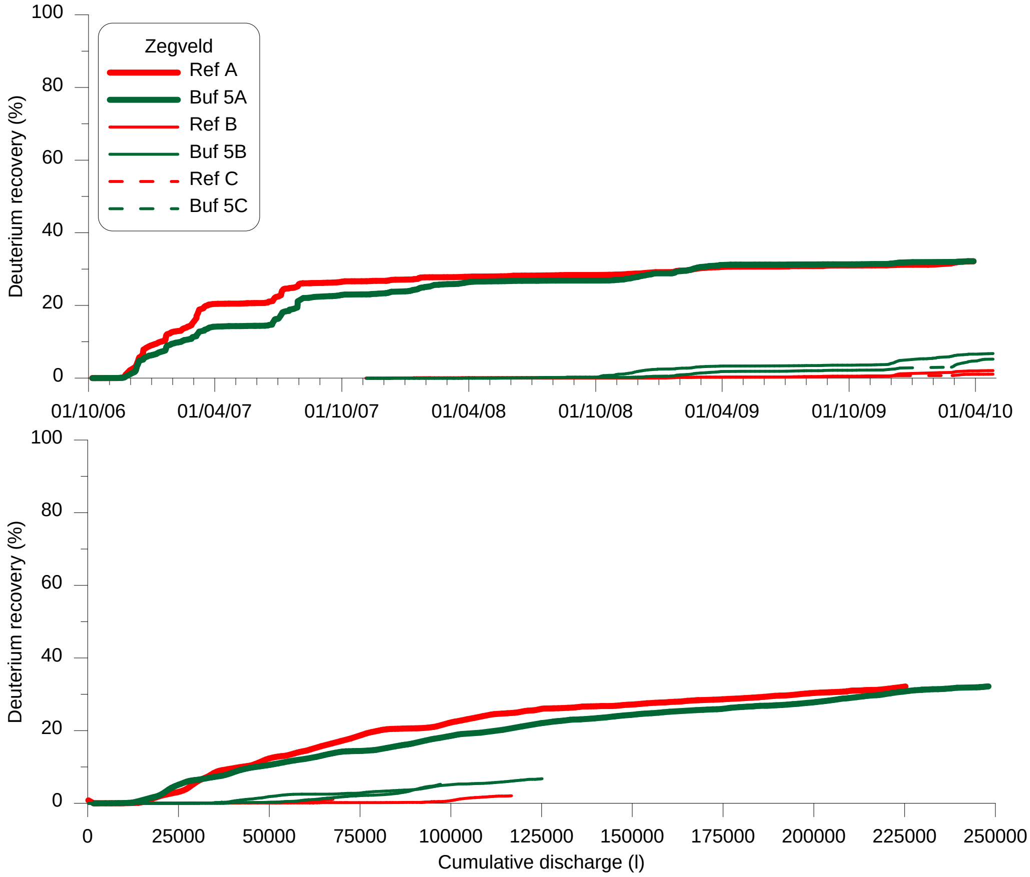


Figure A2.1

Deuterium breakthrough curves for Zegveld

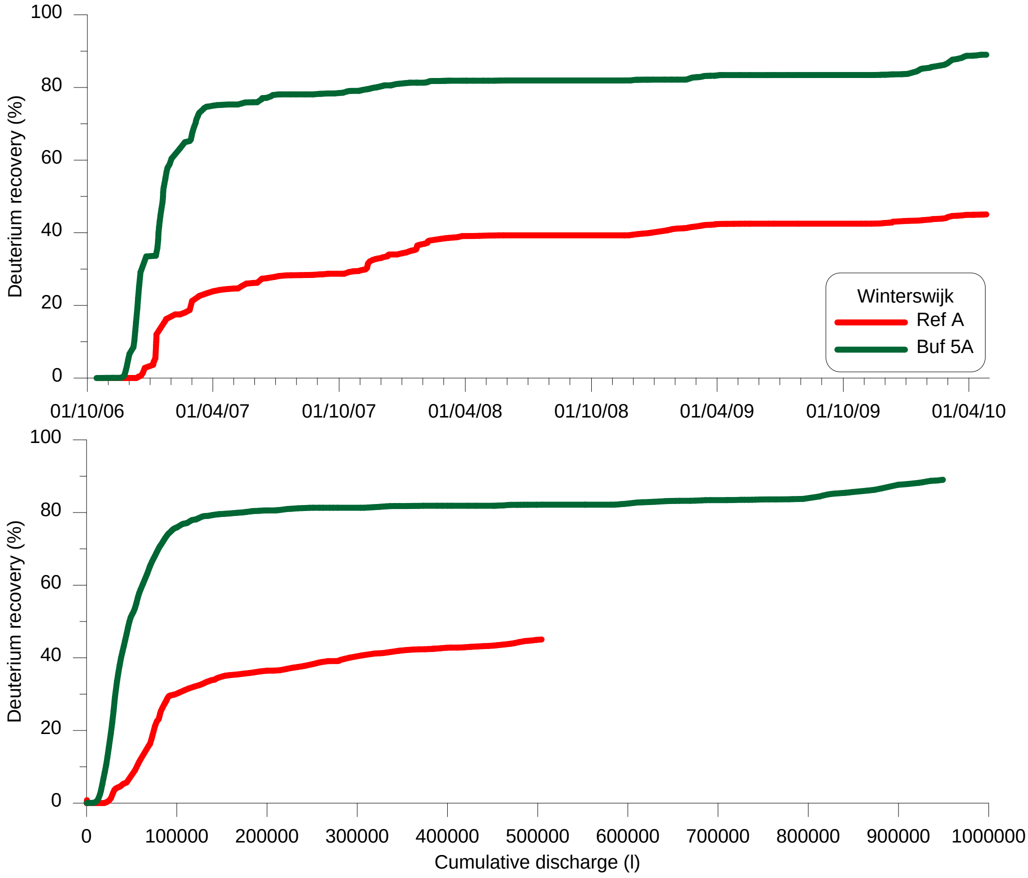


Figure A2.2

Deuterium breakthrough curves for Winterswijk

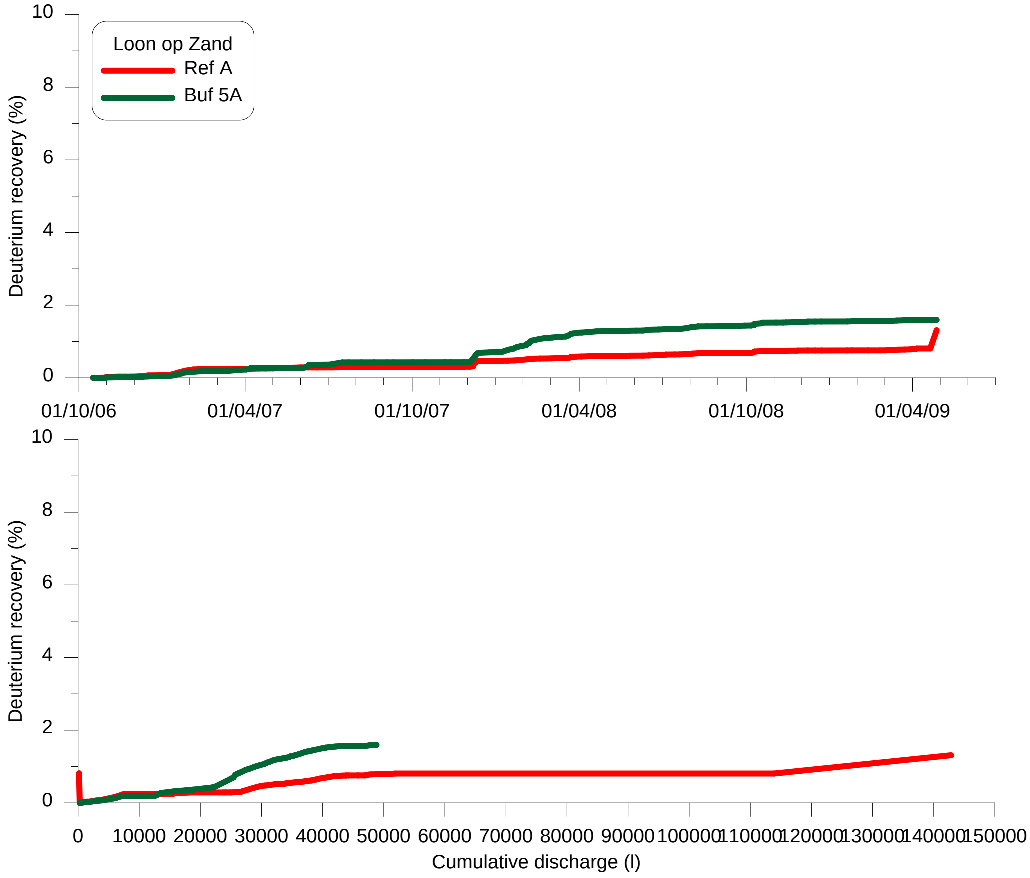


Figure A2.3

Deuterium breakthrough curves for Loon op Zand

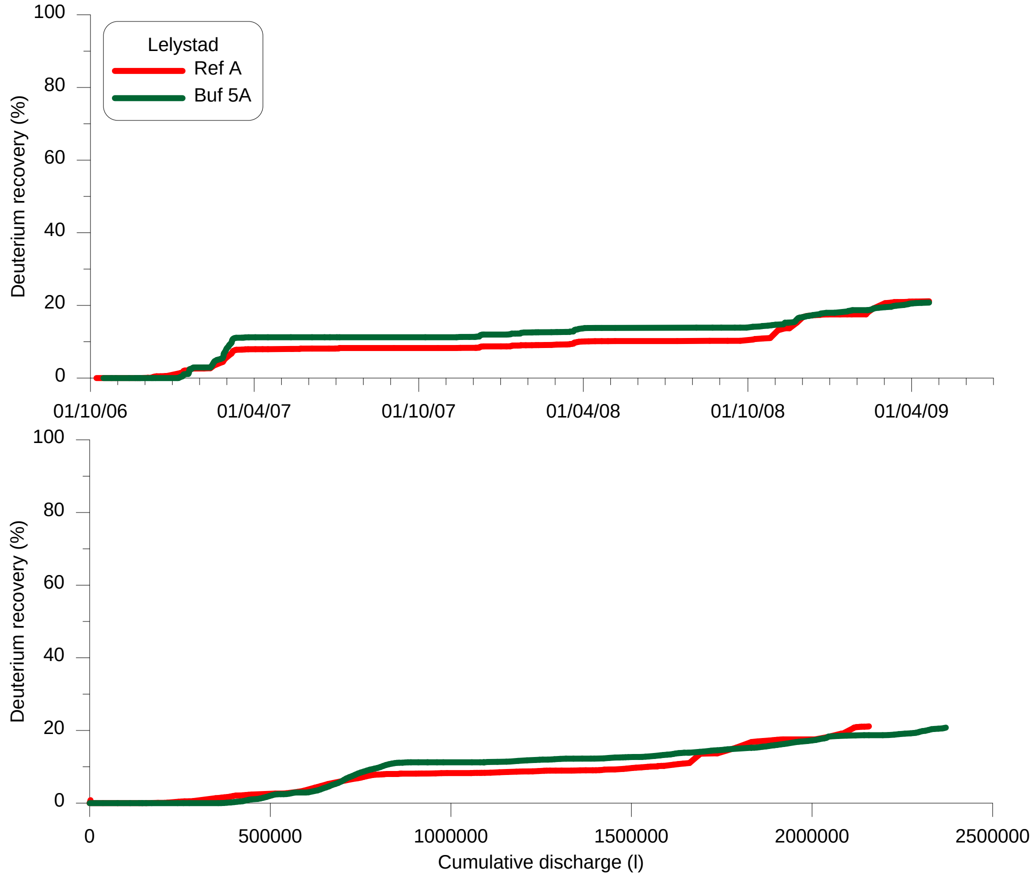


Figure A2.4
Deuterium breakthrough curves for Lelystad

Appendix 3 Concentrations in reservoirs

The measured concentrations in the discharged reservoir water are summarized below in Box-Whisker plots for Cl, DOC, P_{ts}, PO_4, N_{ts}, NO_3, NH_4, and N_{org}, where $N_{org} = N_{ts} – NO_3 – NH_4$. ($N_t$ and P_t were given in the main text). Following these Box-Whisker plots the time courses of N_t, P_t and the above mentioned constituents are given. For comparison, the measured concentrations in the ditch outside the reservoirs is also shown in the time courses.

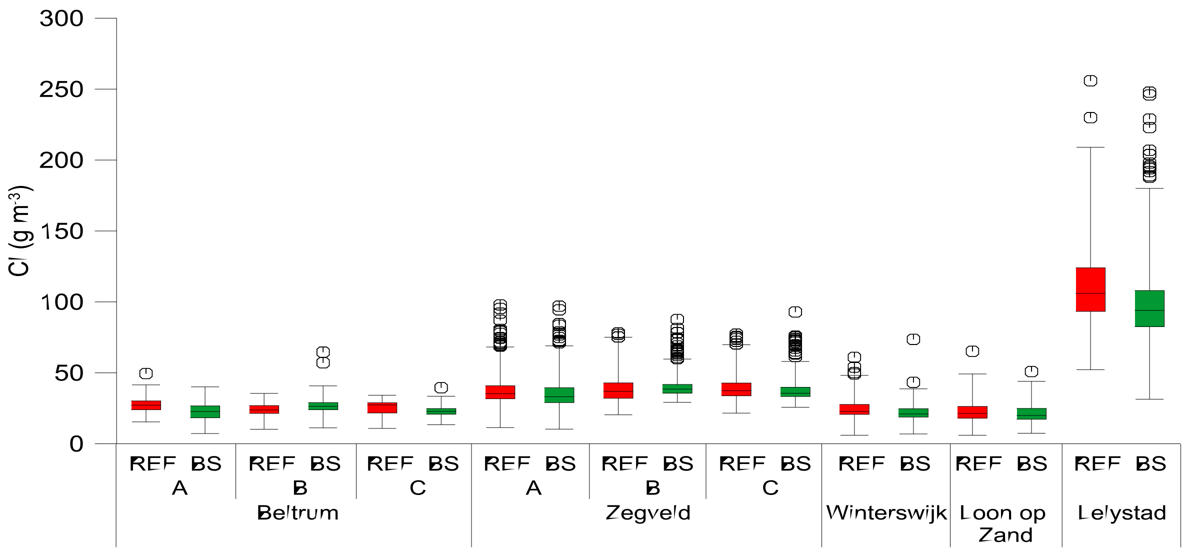


Figure A3.1
Box-whisker plots of the Cl concentrations in the reservoirs at all locations: minimum, first, second (median) and third quartile, and maximum values. Outliers (symbols) are shown which lie more than three times the box length from the box edge

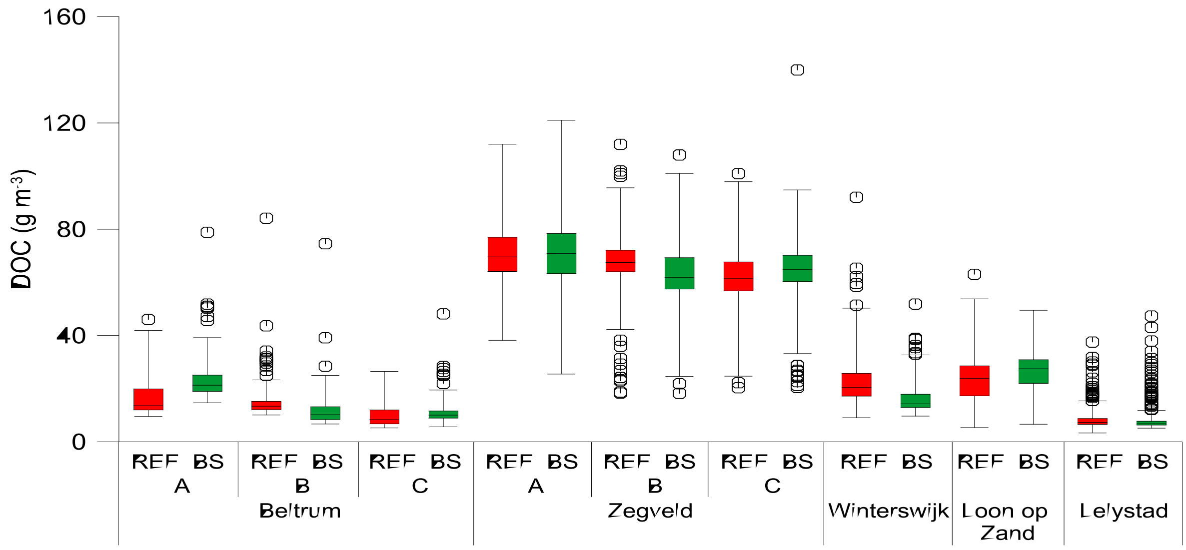


Figure A3.2
Box-whisker plots of the DOC concentrations in the reservoirs at all locations: minimum, first, second (median) and third quartile, and maximum values. Outliers (symbols) are shown which lie more than three times the box length from the box edge

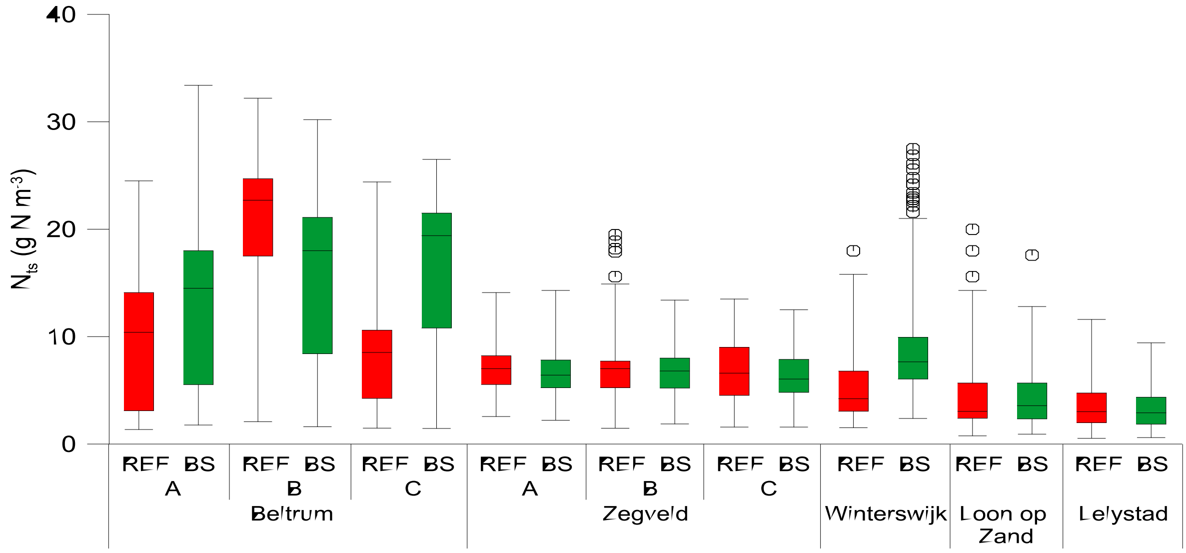


Figure A3.3
Box-whisker plots of the N_{ts} concentrations in the reservoirs at all locations: minimum, first, second (median) and third quartile, and maximum values. Outliers (symbols) are shown which lie more than three times the box length from the box edge

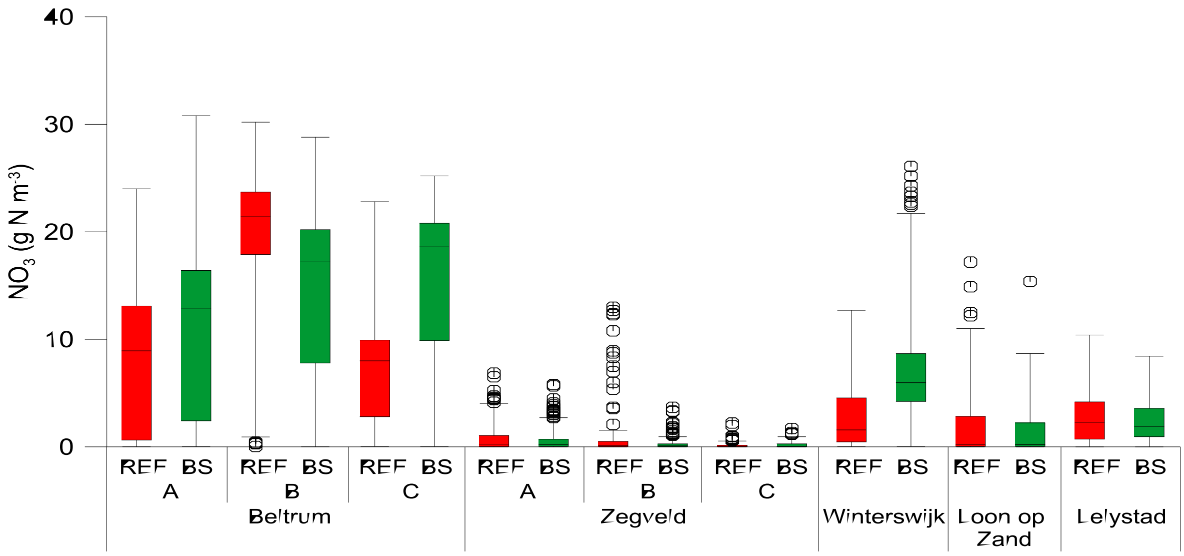


Figure A3.4

Box-whisker plots of the NO_3-N concentrations in the reservoirs at all locations: minimum, first, second (median) and third quartile, and maximum values. Outliers (symbols) are shown which lie more than three times the box length from the box edge

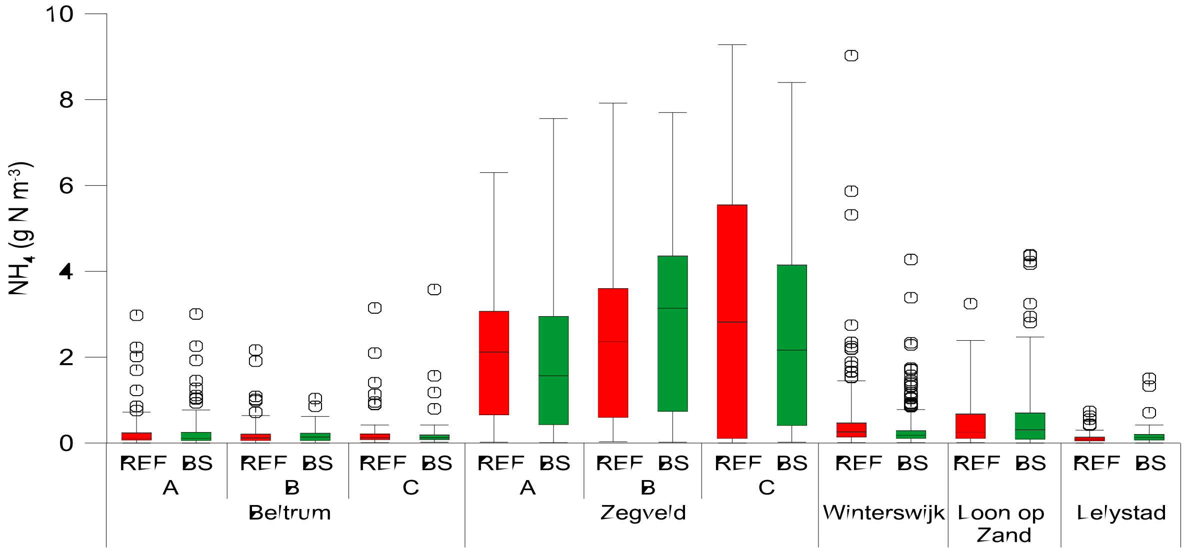


Figure A3.5

Box-whisker plots of the NH_4-N concentrations in the reservoirs at all locations: minimum, first, second (median) and third quartile, and maximum values. Outliers (symbols) are shown which lie more than three times the box length from the box edge

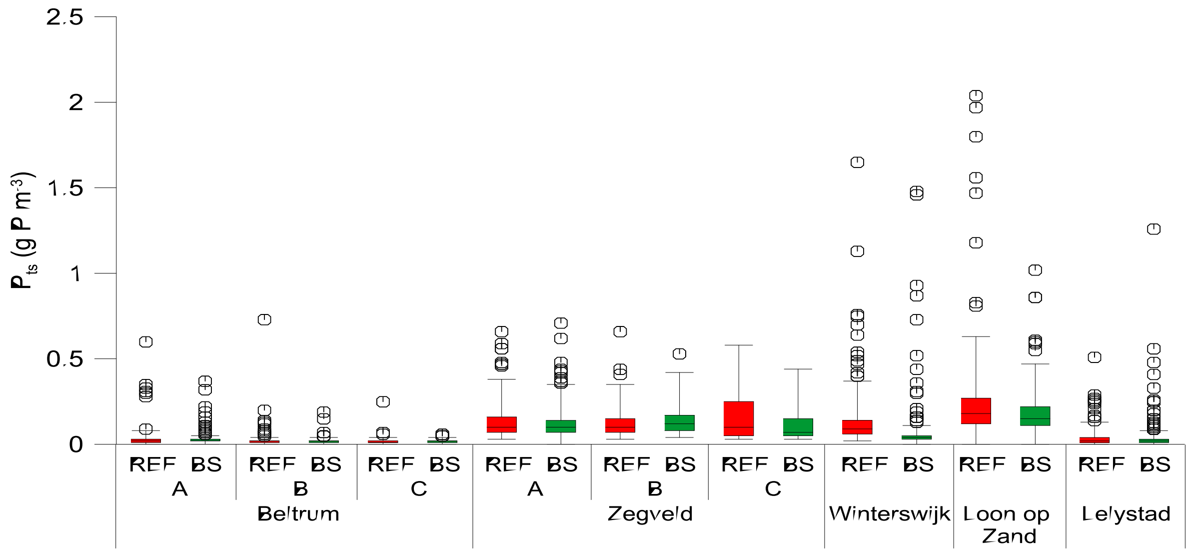


Figure A3.6

Box-whisker plots of the P_{ts} concentrations in the reservoirs at all locations: minimum, first, second (median) and third quartile, and maximum values. Outliers (symbols) are shown which lie more than three times the box length from the box edge

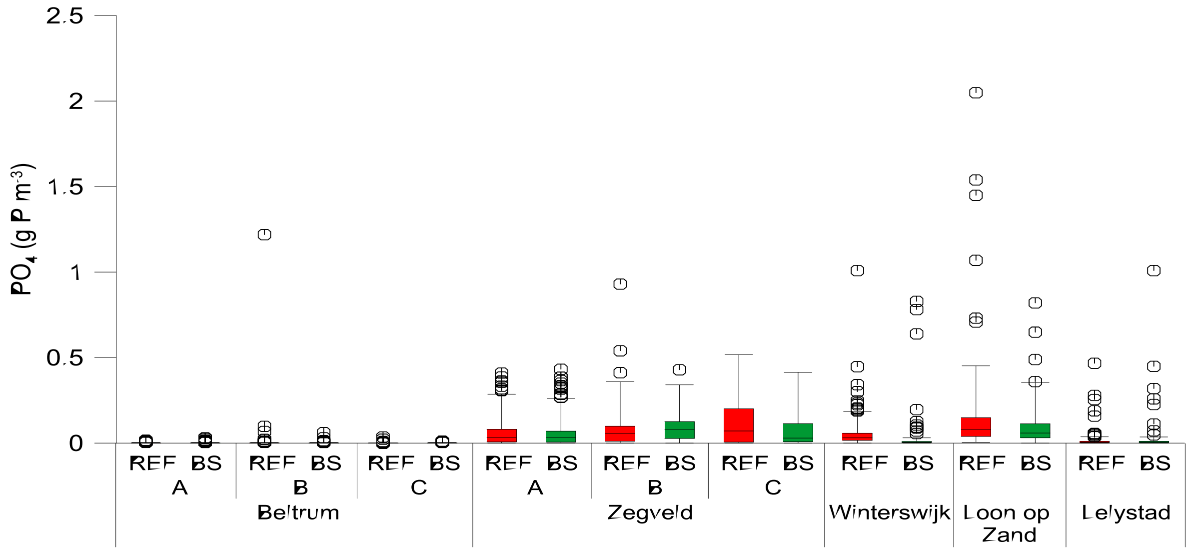


Figure A3.7

Box-whisker plots of the PO_4-P concentrations in the reservoirs at all locations: minimum, first, second (median) and third quartile, and maximum values. Outliers (symbols) are shown which lie more than three times the box length from the box edge

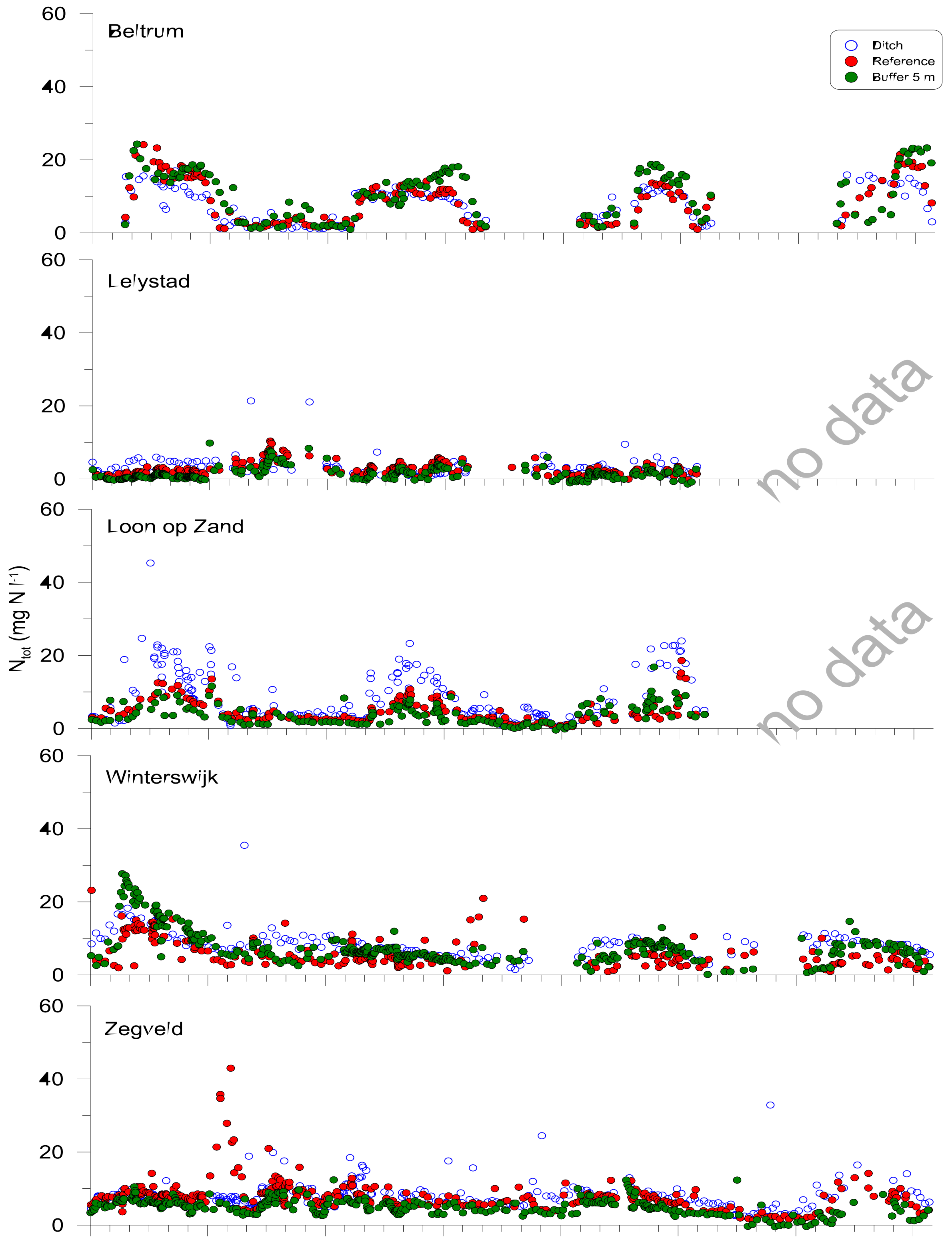


Figure A3.8

Time series of measured N_t concentrations in the BS and REF reservoirs for the five locations (replicate A only). For comparison the concentrations in the ditch outside the reservoirs are given as well

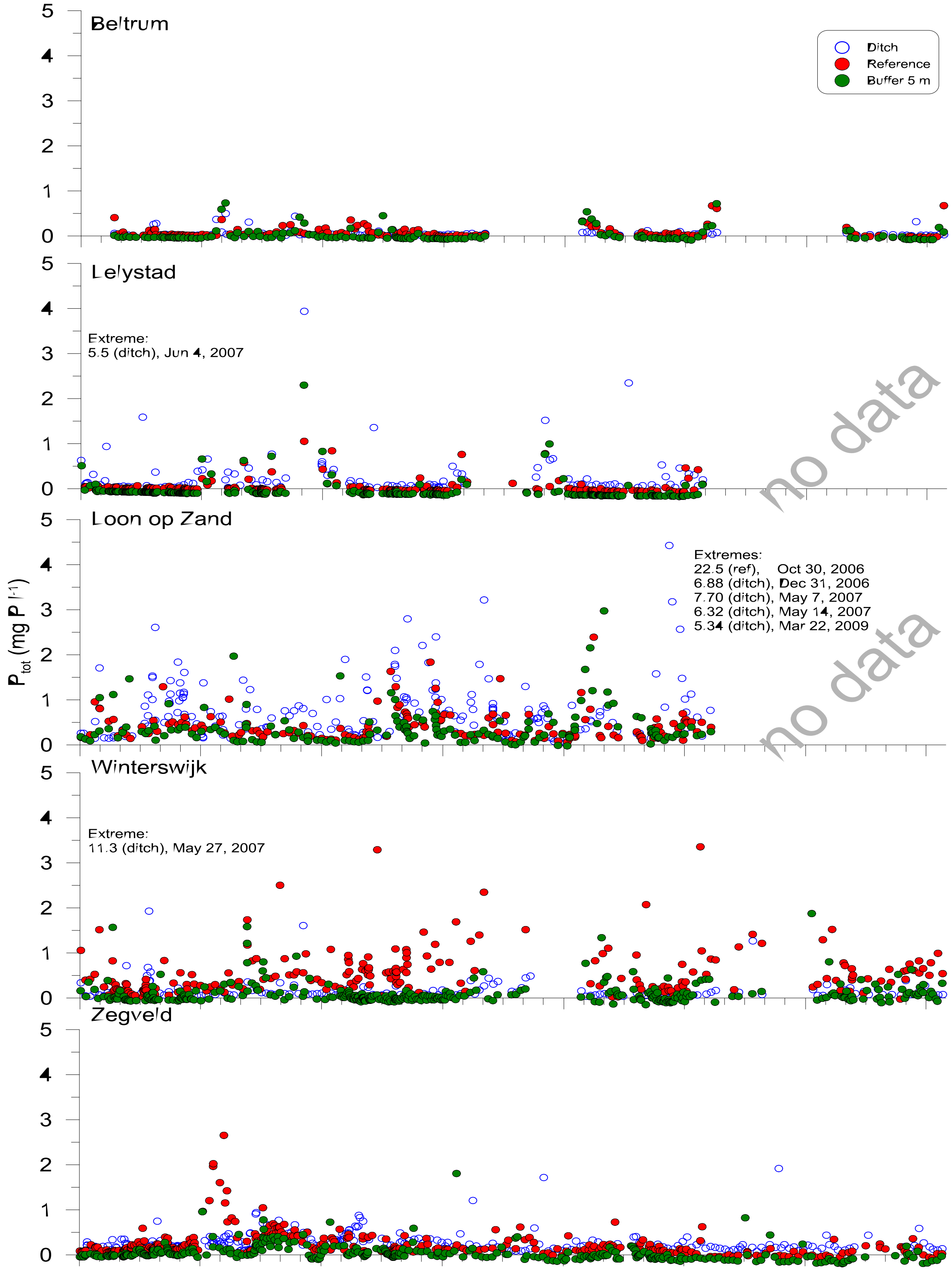


Figure A3.9

Time series of measured P_t concentrations in the BS and REF reservoirs for the five locations (replicate A only). For comparison the concentrations in the ditch outside the reservoirs are given as well

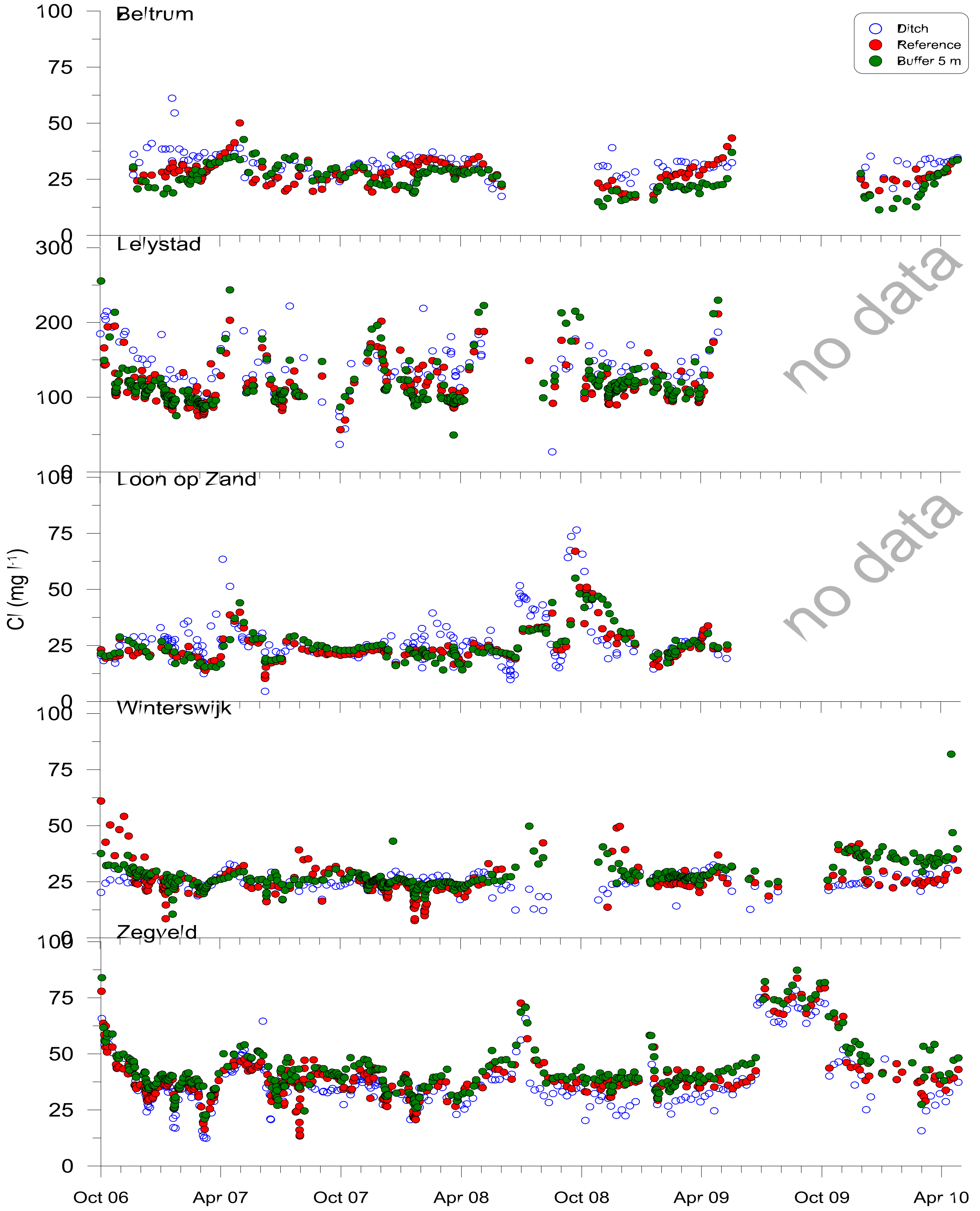


Figure A3.10

Time series of measured Cl concentrations in the BS and REF reservoirs for the five locations (replicate A only). For comparison the concentrations in the ditch outside the reservoirs are given as well

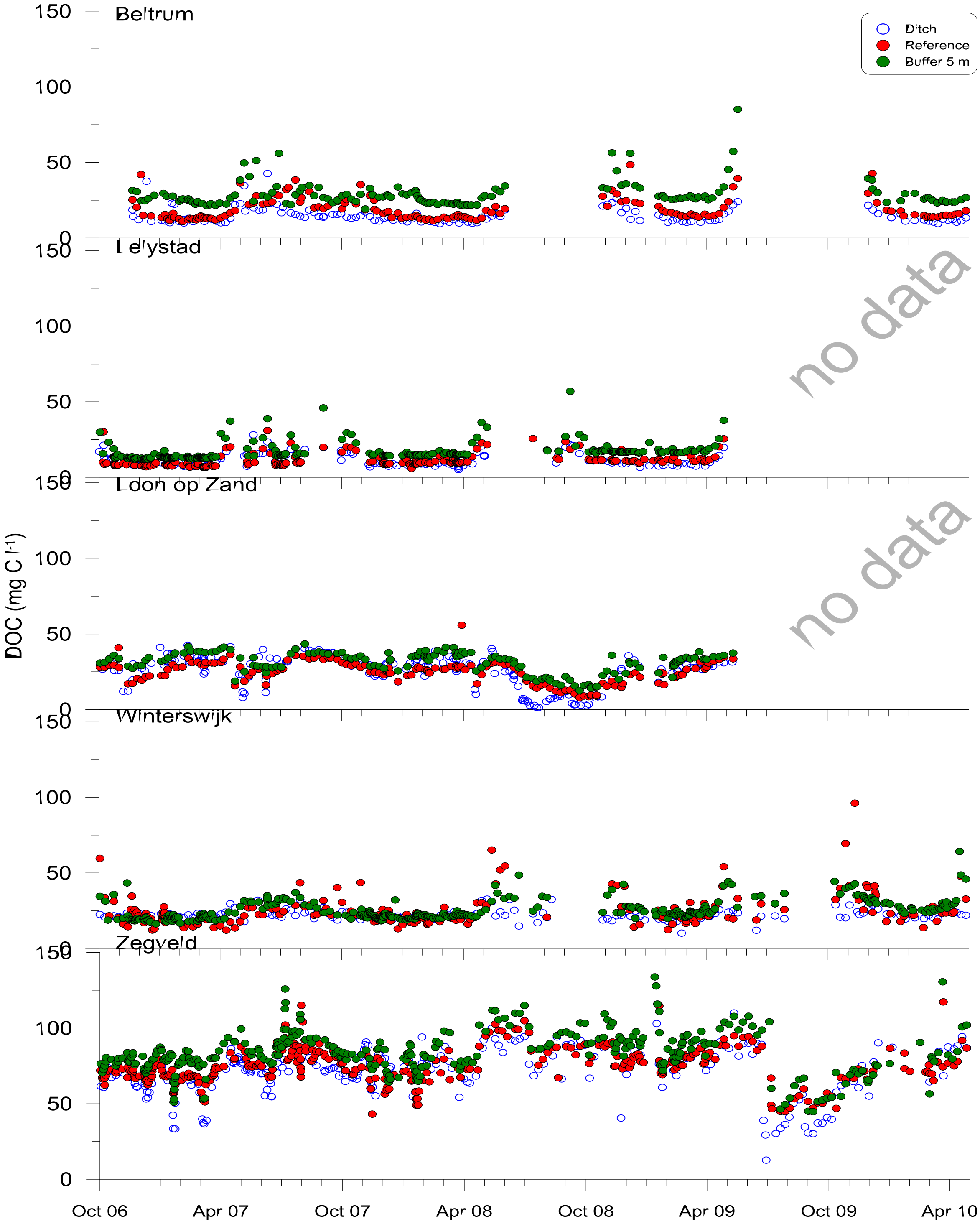


Figure A3.11
Time series of measured DOC concentrations in the BS and REF reservoirs for the five locations (replicate A only). For comparison the concentrations in the ditch outside the reservoirs are given as well

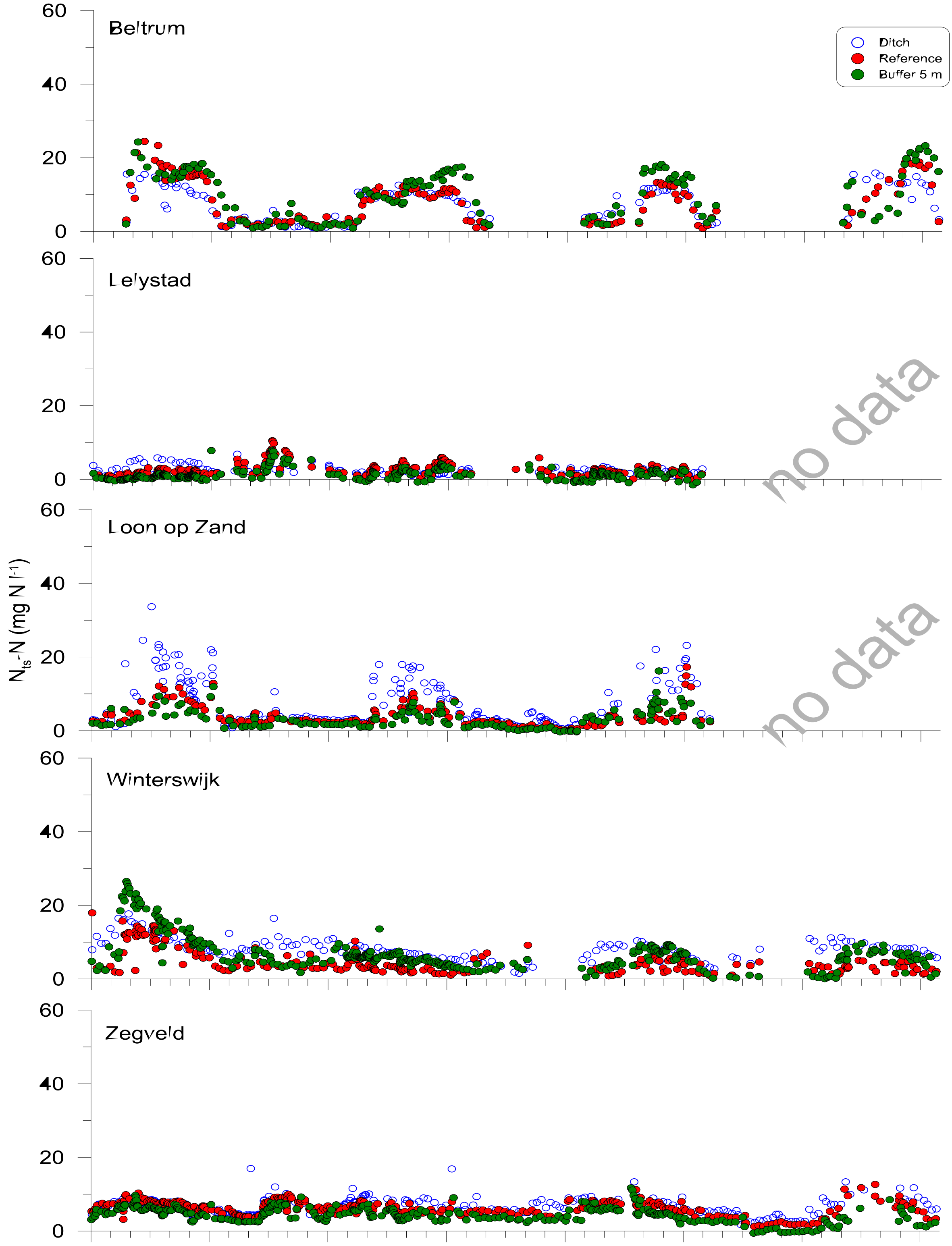


Figure A3.12

Time series of measured N_{ts} concentrations in the BS and REF reservoirs for the five locations (replicate A only). For comparison the concentrations in the ditch outside the reservoirs are given as well

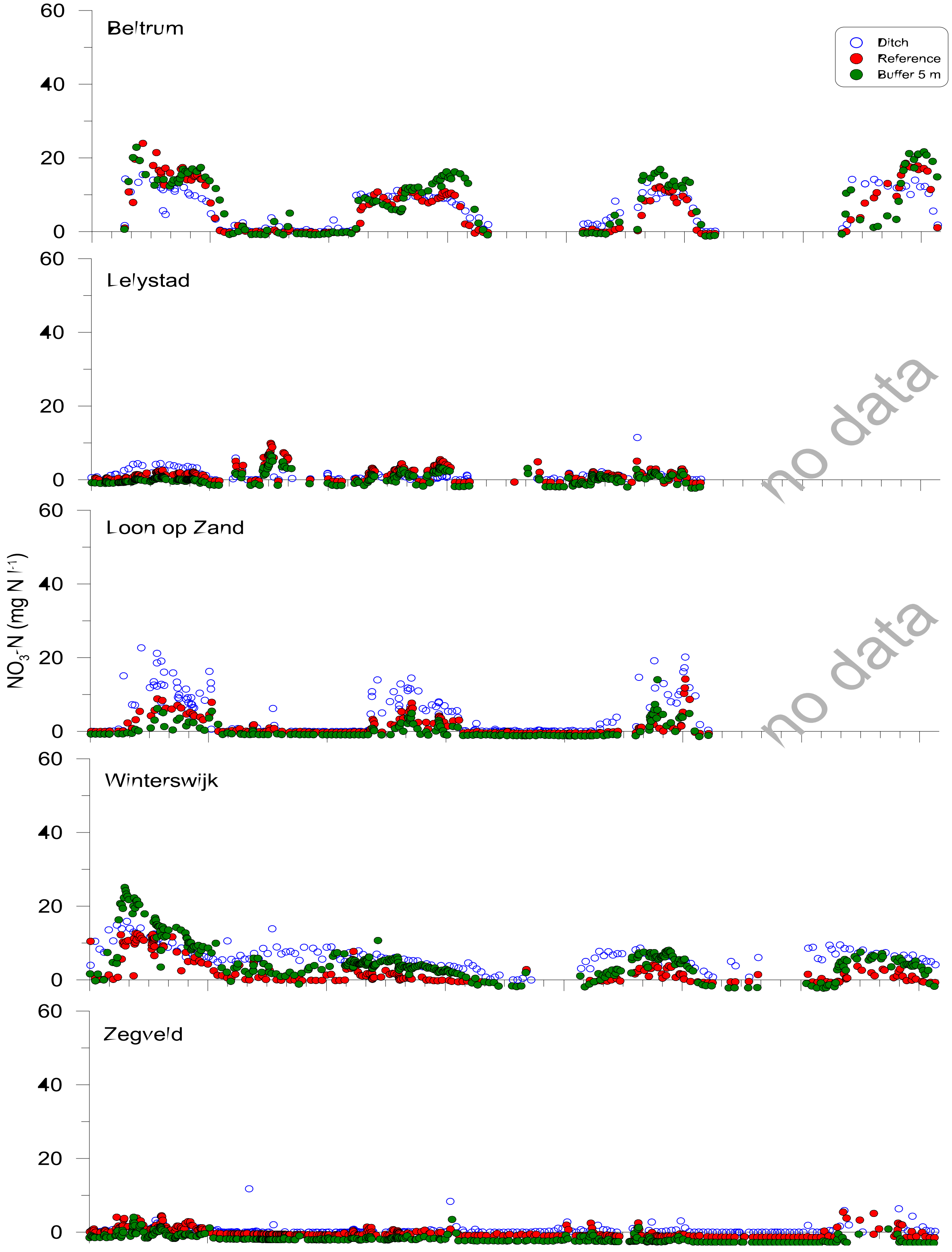


Figure A3.13

Time series of measured NO_3-N concentrations in the BS and REF reservoirs for the five locations (replicate A only). For comparison the concentrations in the ditch outside the reservoirs are given as well

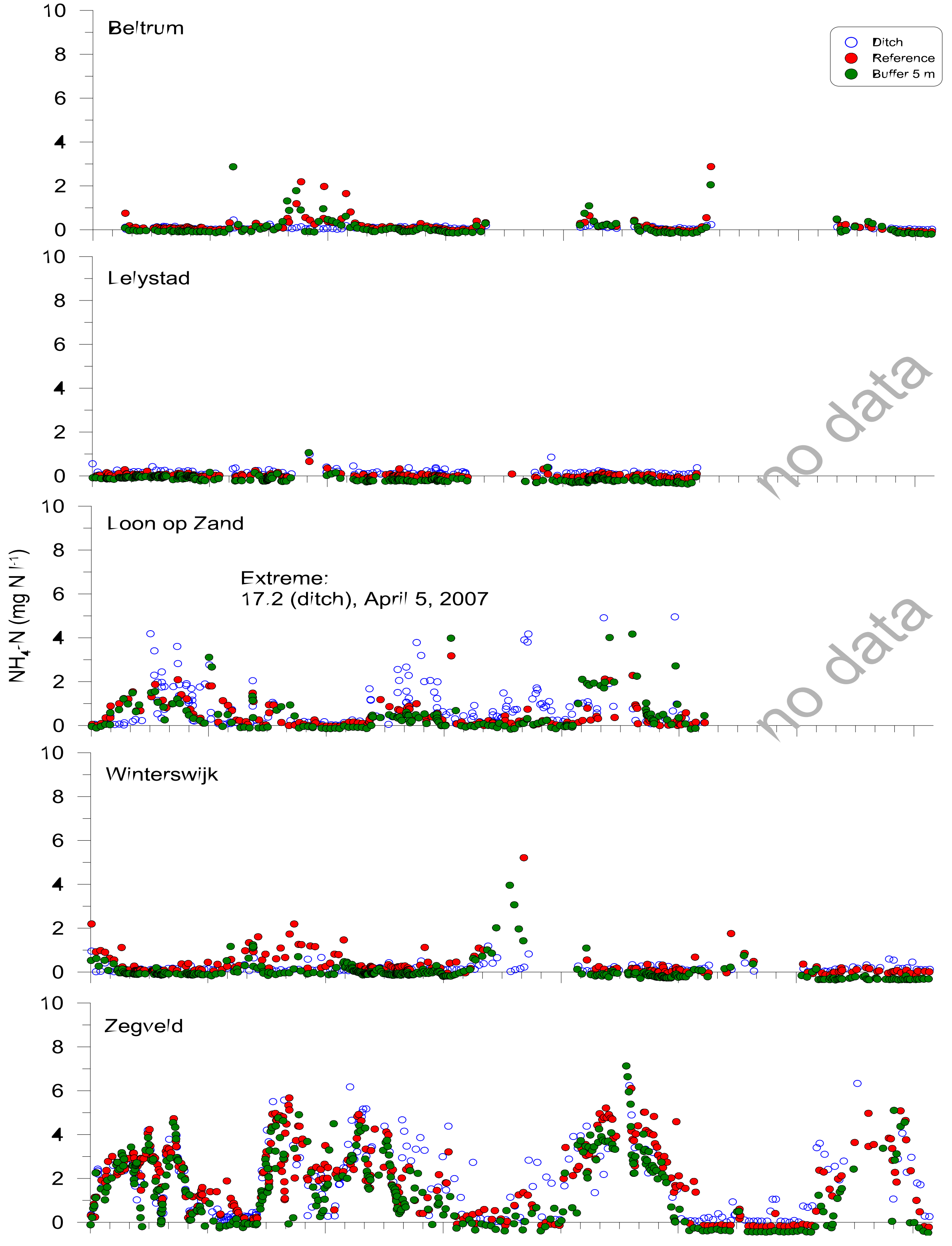


Figure A3.14

Time series of measured NH_4-N concentrations in the BS and REF reservoirs for the five locations (replicate A only). For comparison the concentrations in the ditch outside the reservoirs are given as well

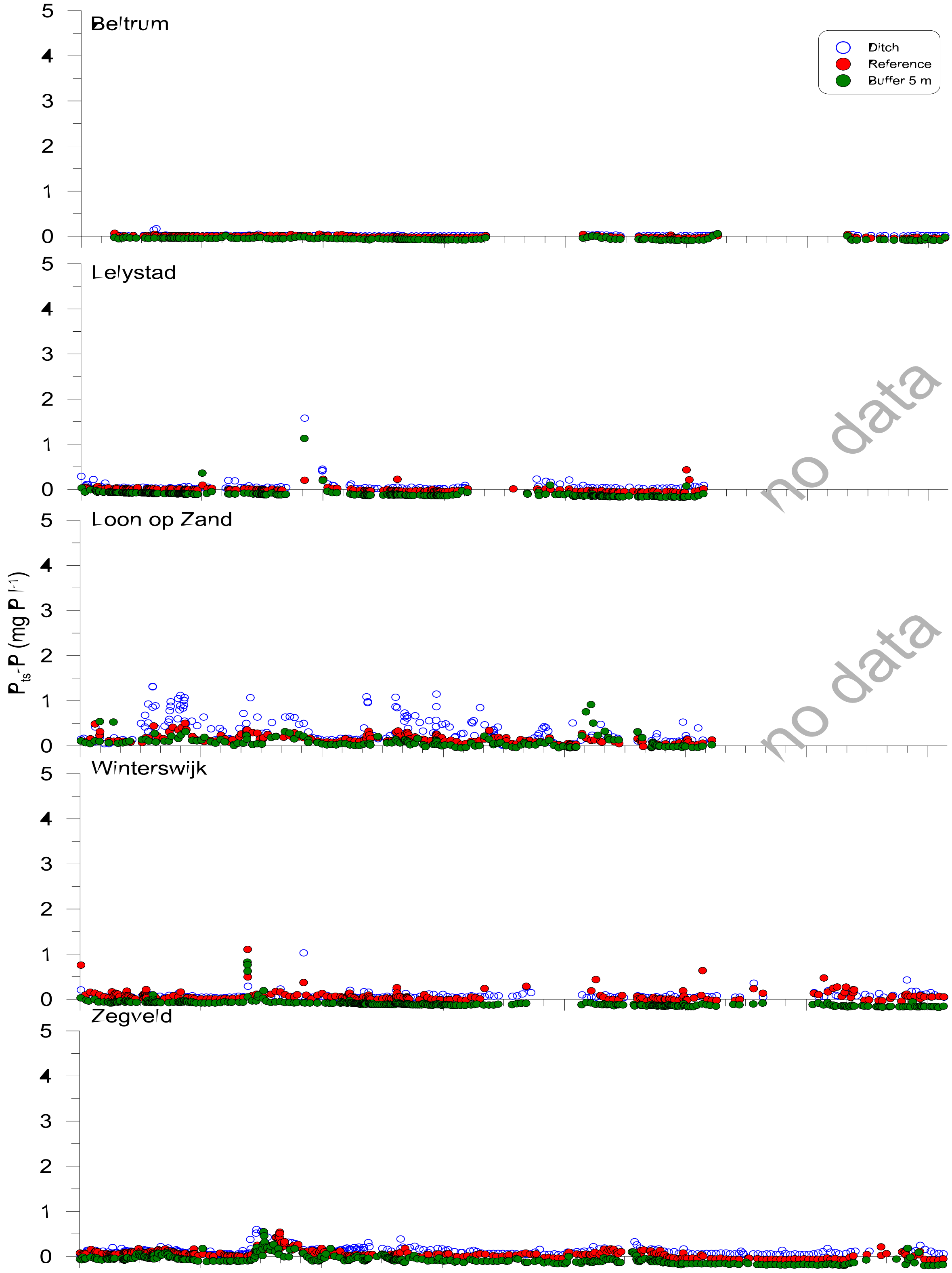


Figure A3.15

Time series of measured P_{ts} concentrations in the BS and REF reservoirs for the five locations (replicate A only). For comparison the concentrations in the ditch outside the reservoirs are given as well

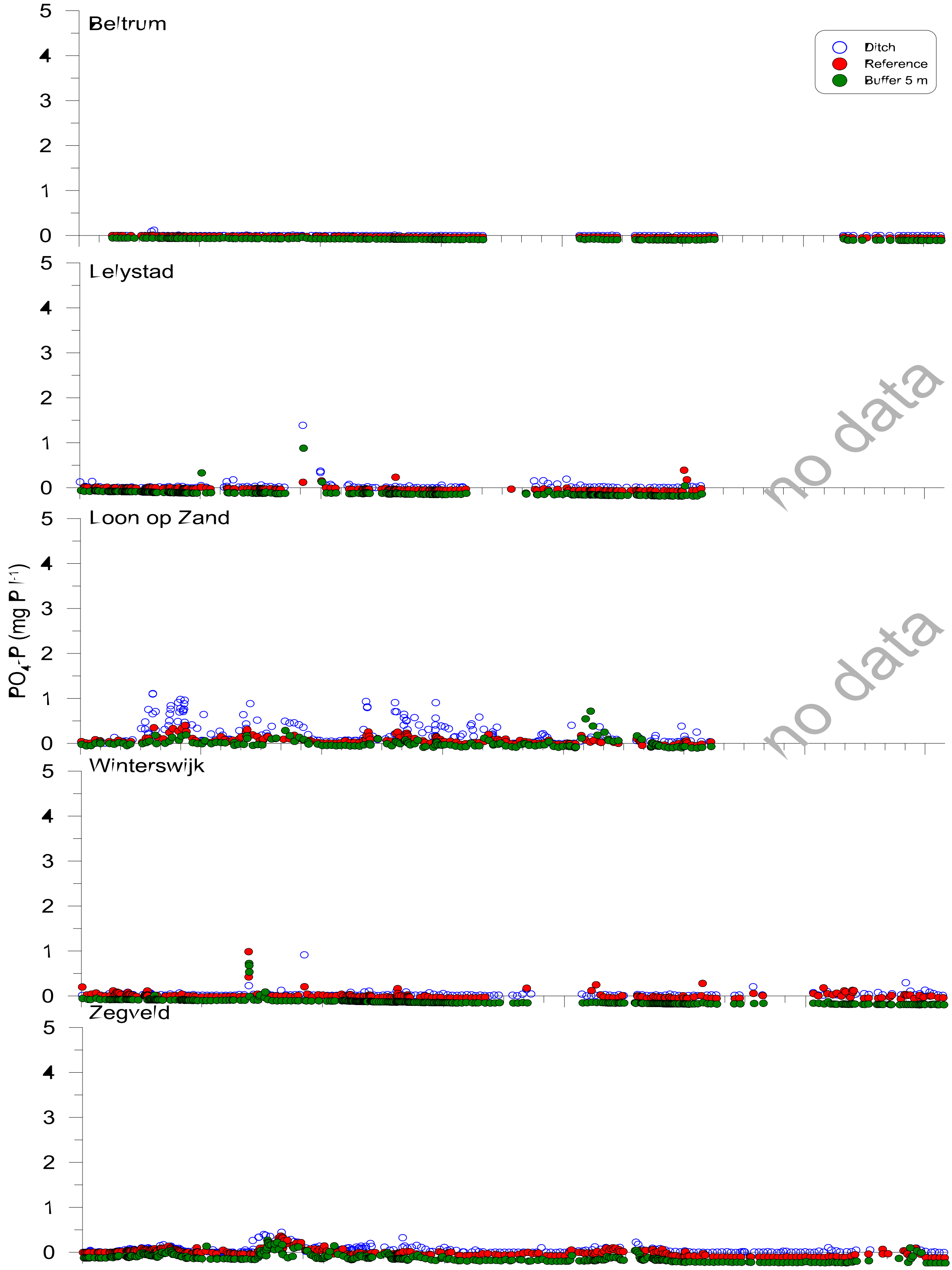


Figure A3.16

Time series of measured PO_4-P concentrations in the BS and REF reservoirs for the five locations (replicate A only). For comparison the concentrations in the ditch outside the reservoirs are given as well

Appendix 4 Concentrations upper groundwater

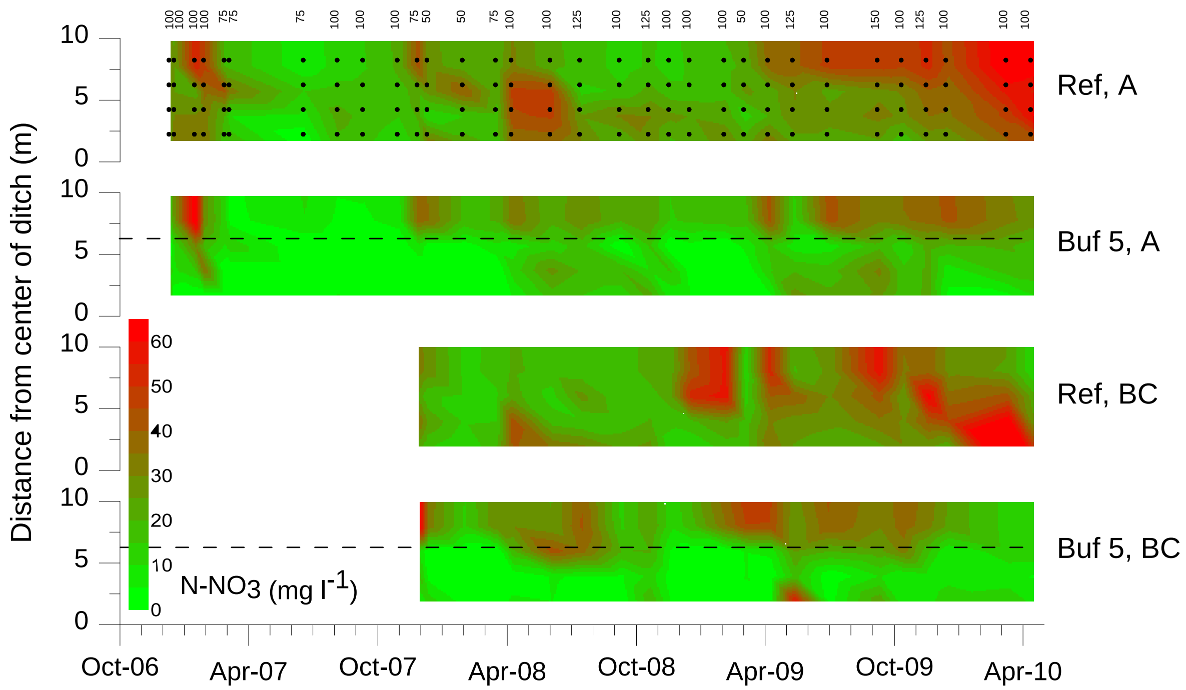


Figure A4.1

Time-contour plots of the nitrate concentration in upper groundwater (mostly at 100 cm bss; indicated at the top) for Beltrum for the BS and REF treatments for replicate A and replicate B+C. The dots represent the sampling points in space and time, and the dotted line represents the outer position of the BS

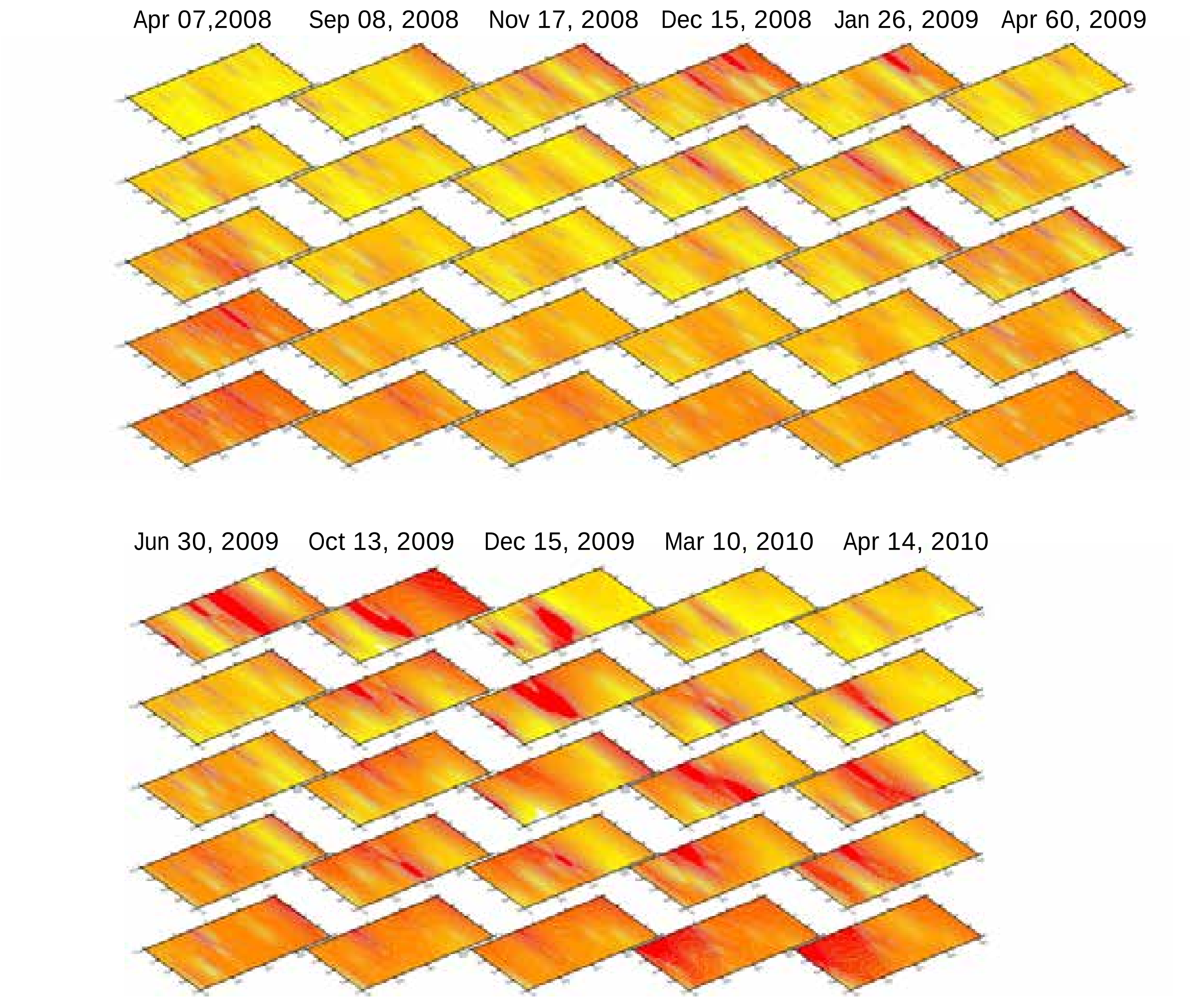


Figure A4.2

Contour plots of the nitrate concentration in upper groundwater at five sampling depths (50, 75, 100, 125, 150 cm bss) for Beltrum A for eleven moments in time when all suction cups were sampled. Yellow indicates zero concentration and full red indicates a concentration of 60 mg L^{-1}

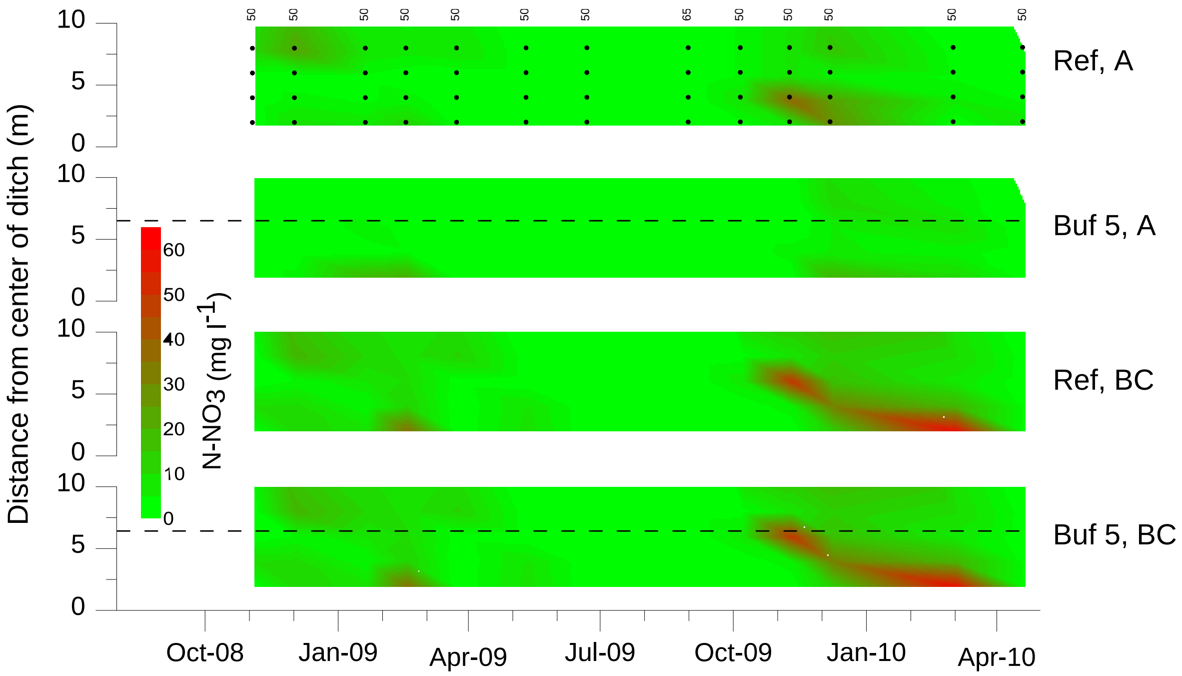


Figure A4.3

Time-contour plots of the nitrate concentration in upper groundwater (at 50 cm bss) for Zegveld for the BS and REF treatments for replicate A and replicate B+C. The dots represent the sampling points in space and time, and the dotted line represents the outer position of the BS

Appendix 5 Concentrations P in upper groundwater

In the Table below the 50 and 90% percentiles of all measured PO_4-P concentrations in upper groundwater are listed.

Table A5.1
Effect of buffer strip on upper groundwater PO_4-P concentration (C_{gw}) for treatments REF and BS and their difference. Grey cells indicate a concentration below the detection limit (0.02 g m^3)

Site	$C_{gw,}$ 50 percentile			$C_{gw,}$ 90 percentile		
	REF	BS	REF-BS	REF	BS	REF-BS
Beltrum	0.001	0.001	0.000	0.005	0.011	-0.006
Zegveld	0.002	0.002	0.000	0.019	0.009	0.009
Winterswijk	0.045	0.006	0.039	0.069	0.055	0.014
Loon op Zand	0.004	0.002	0.002	0.030	0.008	0.022
Lelystad	0.019	-0.001	0.020	0.042	0.018	0.023

Appendix 6 Analytical expressions for the travel time as a function of the distance to the ditch

For the hydrogeological schematization a phreatic aquifer is supposed to convey the precipitation surplus from field to ditch (3rd order) or canal (2nd order). This conceptual model for the relation between discharge rates and resp. recharge, surface water level and hydraulic head of deep groundwater was used by De Lange (1999) for deriving of so-called "phreatic seep resistances" and has been implemented in the National Hydrologic Model for the Netherlands. It supposes that a phreatic aquifer can leak water through a semi-permeable aquitard with resistance c_1 under which a certain hydraulic head ϕ is maintained by a regional groundwater flow system (Fig. A6.1). The bottom of the ditch is semi-permeable and a certain surface water level p is maintained.

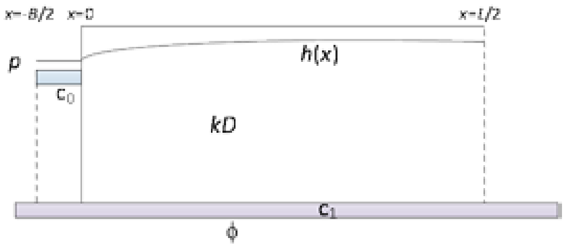


Figure A6.1
Schematic representation of the flow domain considered

The hydraulic head of the phreatic aquifer in the land system $h_1(x)$ is a function of the distance to the interface between land and ditch:

$$h_1(x) = Rc_1 + \varphi + \left(p - R(c_0 + c_1) - \varphi\right) \frac{\dfrac{c_1}{c_0 + c_1}\lambda_L}{\lambda_B \coth\left(\dfrac{B}{2\lambda_B}\right) + \lambda_L \coth\left(\dfrac{L}{2\lambda_L}\right)} \frac{\cosh\left(\dfrac{L/2 - x}{\lambda_L}\right)}{\sinh\left(\dfrac{L/2}{\lambda_L}\right)} \qquad \text{[A6.1]}$$

where $L/2$ is the distance between the ditch and the water divide, $B/2$ is half of the ditch width, R is the recharge, p is the surface water level, φ is the hydraulic head of the regional groundwater flow system, c_1 is the resistance of the semi-permeable layer beneath the aquifer, c_0 is the resistance of the ditch bottom. The characteristic lengths λ_L and λ_B are calculated as $\lambda_L = \sqrt{c_1 kD}$ and $\lambda_B = \sqrt{kD \dfrac{c_0 c_1}{c_0 + c_1}}$ where kD is the transmissivity of the aquifer.

The hydraulic head in the aquifer below the ditch $h_2(x)$ reads as:

$$h_2(x) = \frac{\varphi c_0 + p c_1}{c_0 + c_1} - \left(p - R(c_0 + c_1) - \varphi\right) \frac{\dfrac{c_1}{c_0 + c_1}\lambda_B}{\lambda_B \coth\left(\dfrac{B}{2\lambda_B}\right) + \lambda_L \coth\left(\dfrac{L}{2\lambda_L}\right)} \frac{\cosh\left(\dfrac{B/2 + x}{\lambda_B}\right)}{\sinh\left(\dfrac{B/2}{\lambda_B}\right)} \qquad \text{[A6.2]}$$

The horizontal flow rate as a function of the distance in the aquifer of resp. the land system and below the ditch:

$$Q_1(x) = -\left(R + \frac{\varphi - p}{c_0 + c_1}\right) \frac{\lambda_L^2}{\lambda_B \coth\left(\dfrac{B}{2\lambda_B}\right) + \lambda_L \coth\left(\dfrac{L}{2\lambda_L}\right)} \frac{\sinh\left(\dfrac{L/2 - x}{\lambda_L}\right)}{\sinh\left(\dfrac{L/2}{\lambda_L}\right)} \qquad \text{[A6.3]}$$

$$Q_2(x) = -\left(R + \frac{\varphi - p}{c_0 + c_1}\right) \frac{\lambda_L^2}{\lambda_B \coth\left(\dfrac{B}{2\lambda_B}\right) + \lambda_L \coth\left(\dfrac{L}{2\lambda_L}\right)} \frac{\sinh\left(\dfrac{B/2 - x}{\lambda_B}\right)}{\sinh\left(\dfrac{B/2}{\lambda_B}\right)} \qquad \text{[A6.4]}$$

For $Q(0)$ holds:

$$Q(0) = -\left(R + \frac{\varphi - p}{c_0 + c_1}\right) \frac{\lambda_L^2}{\lambda_B \coth\left(\dfrac{B}{2\lambda_B}\right) + \lambda_L \coth\left(\dfrac{L}{2\lambda_L}\right)} \qquad \text{[A6.5]}$$

From which it follows:

$$Q_1(x) = Q(0) \frac{\sinh\left(\dfrac{L/2 - x}{\lambda_L}\right)}{\sinh\left(\dfrac{L/2}{\lambda_L}\right)} \text{ and } Q_2(x) = Q(0) \frac{\sinh\left(\dfrac{B/2 + x}{\lambda_B}\right)}{\sinh\left(\dfrac{B/2}{\lambda_B}\right)} \qquad \text{[A6.6]}$$

The travel time is calculated from:

$$\varepsilon \frac{\mathrm{d}x}{\mathrm{d}t} = -\frac{Q(x)}{H} \rightarrow t_{x1} - t_{x0} = -\int_{x0}^{x1} \frac{\varepsilon H}{Q(x)} \mathrm{d}x \qquad \text{[A6.7]}$$

Where ε is the porosity and t is time of a parcel of water after infiltration. Subscripts denote the position of the water parcel

For $x > 0$ the travel time as a function of the distance to the interface between land system and ditch at $x = 0$ is given by:

$$t_L(x) = -\frac{\varepsilon H}{Q(0)} \lambda_L \sinh\left(\frac{L/2}{2\lambda_L}\right) \ln\left(\frac{\coth\left(\dfrac{L/2 - x}{2\lambda_L}\right)}{\coth\left(\dfrac{L/2}{2\lambda_L}\right)}\right) \qquad \text{[A6.8]}$$

And for $x < 0$ the travel time from the interface between the land system and the ditch to the exfiltration point at x is calculated by:

$$t_B(x) = -\frac{\varepsilon H}{Q(0)} \lambda_B \sinh\left(\frac{B/2}{2\lambda_B}\right) \ln\left(\frac{\coth\left(\frac{B/2 + x}{2\lambda_B}\right)}{\coth\left(\frac{B/2}{2\lambda_B}\right)}\right) \qquad \text{[A6.9]}$$

Some special cases can be considered:

1. Ditch width is very small compared to the distance between ditches

B and c_0 are set to zero and consequently λ_B equals zero. Then for $Q(0)$ holds:

$$Q(0) = -\left(R + \frac{\varphi - \pi}{c_1}\right)\lambda_L \tanh\left(\frac{L}{2\lambda_L}\right) \qquad \text{[A6.10]}$$

The travel time $t_L(x)$ is calculated by using Eq. [A6.8] and the travel time $t_B(x)$ is set to zero.

2. Resistance of semi-permeable aquitard is very high and closes the bottom boundary

The flow rate as a function of x is given by:

$$Q(x) = -R\left(\frac{L}{2} - x\right) \qquad \text{[A6.11]}$$

Which yields the following expression for the travel time:

$$t_L(x) = \frac{\varepsilon H}{R} \ln\left(\frac{L/2}{L/2 - x}\right) \qquad \text{[A6.12]}$$

3. Both resistance of semi-permeable aquitard is very high and the hydraulic head difference over this layer is relatively high, which results in a more or less uniformly distributed upward or downward seepage flux over this layer.

When both the resistance of semi-permeable aquitard is very high and the hydraulic head difference over this layer is relatively high a more or less uniformly distributed upward or downward seepage flux will occur over this layer. We distinguish a situation with upward seepage and a situation with downward seepage (S<0) (Fig. A6.2).

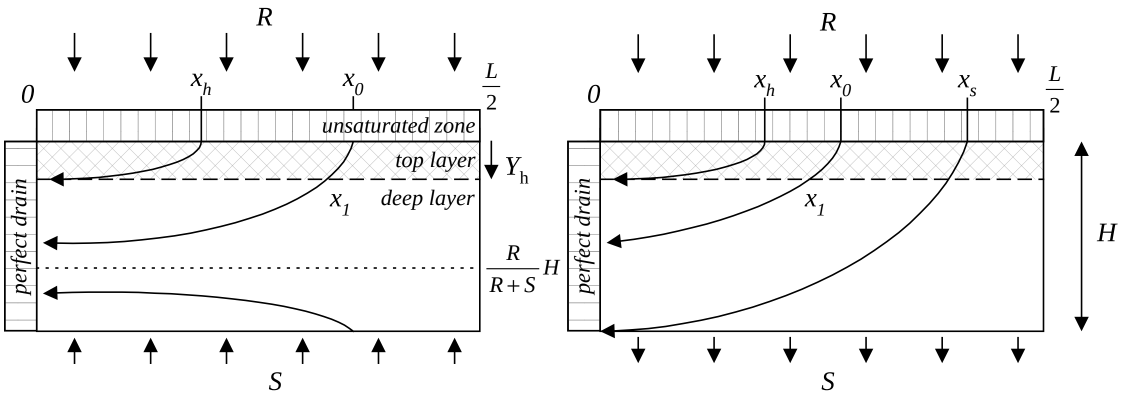


Figure A6.2
Schematization of the flow domain drained by fully penetrating drain in case of an upward seepage flux (left) and in case of a downward seepage flux (right)

Distinguishing biogeochemical subdomains
The strip along a water course is often wetter than the remainder of the field resulting in higher potential denitrification rates. Also, the top soil contains more fresh organic matter than the subsoil. Therefore, different biogeochemical subdomains are distinguished with potentially different reaction rate constants k (Fig. A6.3). The width of the wet zone adjacent to the ditch is set to x_R and the top soil with higher reactivity has a depth h.

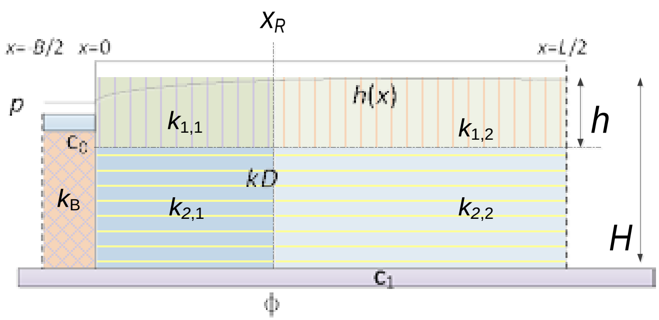


Figure A6.3
Characterization of biogeochemical subdomains with different first order rate constants k

To calculate the concentration in the flow towards the ditch, the travel time of a parcel of water on a stream line should be established for each subdomain. This requires an algebraic expression with the coordinates x, y for the stream lines. There is no such general algebraic solution , but expressions can be found based on flow rate proportions :

- For $0 < x < L/2$: The height y of a point on a streamline that starts in at a certain position X_1, follows from the water volume that passes between H and H-y and the extend of the infiltration zone that corresponds to the water volume :

$$-Q_1(x)\frac{H-y}{H} = R(X_1 - x) \qquad [A6.13]$$

- For $-B/2 < x < 0$: The height y of a point on a streamline that ends at a certain position X_2 follows from the water volume that passes between H and H-y and the extend of the exfiltration zone that corresponds to the water volume :

$$-Q_2(x)\frac{H-y}{H} = -\int_{X_2}^{x} \frac{h_2(x)-p}{c_0}\,\mathrm{d}x \qquad [A6.14]$$

Appendix 7 Travel time as a function of the distance for converging and diverging flow systems

The special case of converging or diverging flow requires an adaptation of the travel time relations as a function of the distance. We need to distinguish between diverging / converging flow seen from the top view position and diverging / converging flow seen from the side view position (Fig.A7.1)

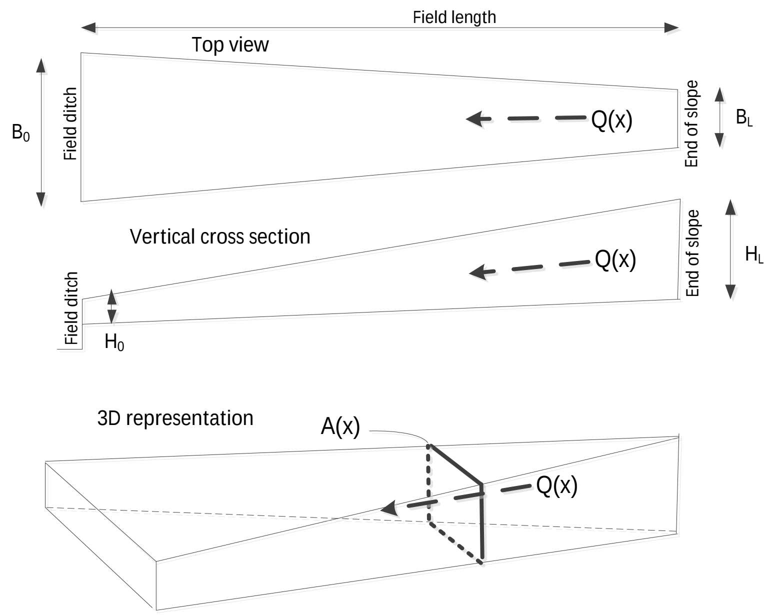


Fig. A7.1
Simplified geometry of a discharging area of a field with diverging flow from the top view position and converging flow from the side view position

The relation for the travel time as a function of distance is derived from:

$$e\frac{\mathrm{d}x}{\mathrm{d}t} = -\frac{Q(x)}{A(x)} \qquad [A7.1]$$

Where $Q(x)$ is the discharge through a plane with area $A(x)$ which dissects the groundwater volume at distance x. The discharge $Q(x)$ and the area $A(x)$ for this special case can be expressed as:

$$Q(x) = -R\left\{(L - x)B_0 + 0.5p_1\left(L^2 - x^2\right)\right\} \qquad [A7.2]$$

$$A(x) = \left(B_0 + p_1 x\right)\left(H_0 + p_2 x\right) \qquad [A7.3]$$

L is the distance between field ditch and water divide, B_0 is the length of the strip subject to evaluation of buffer strips (=12,5 m), H_0 the height of the groundwater body adjacent to the ditch and p_1 and p_2 are shape parameters. The resulting equation for the travel time as a function of the distance to the ditch:

$$t(x) = \frac{\varepsilon}{R}\left\{ (H_0 + Lp_2)\ln\left(\frac{L}{L-x}\right) + \left(2B_0\frac{p_2}{p_1} + Lp_2 - H_0\right)\ln\left(\frac{2B_0 + p_1(L+x)}{2B_0 + Lp_1}\right) - 2p_2 x \right\} \quad \text{[A7.4]}$$

For the special case of $p_1 = 0$:

$$t(x) = \frac{\varepsilon}{R}\left\{ (H_0 + Lp_2)\ln\left(\frac{L}{L-x}\right) - p_2 x \right\} \quad \text{[A7.5]}$$

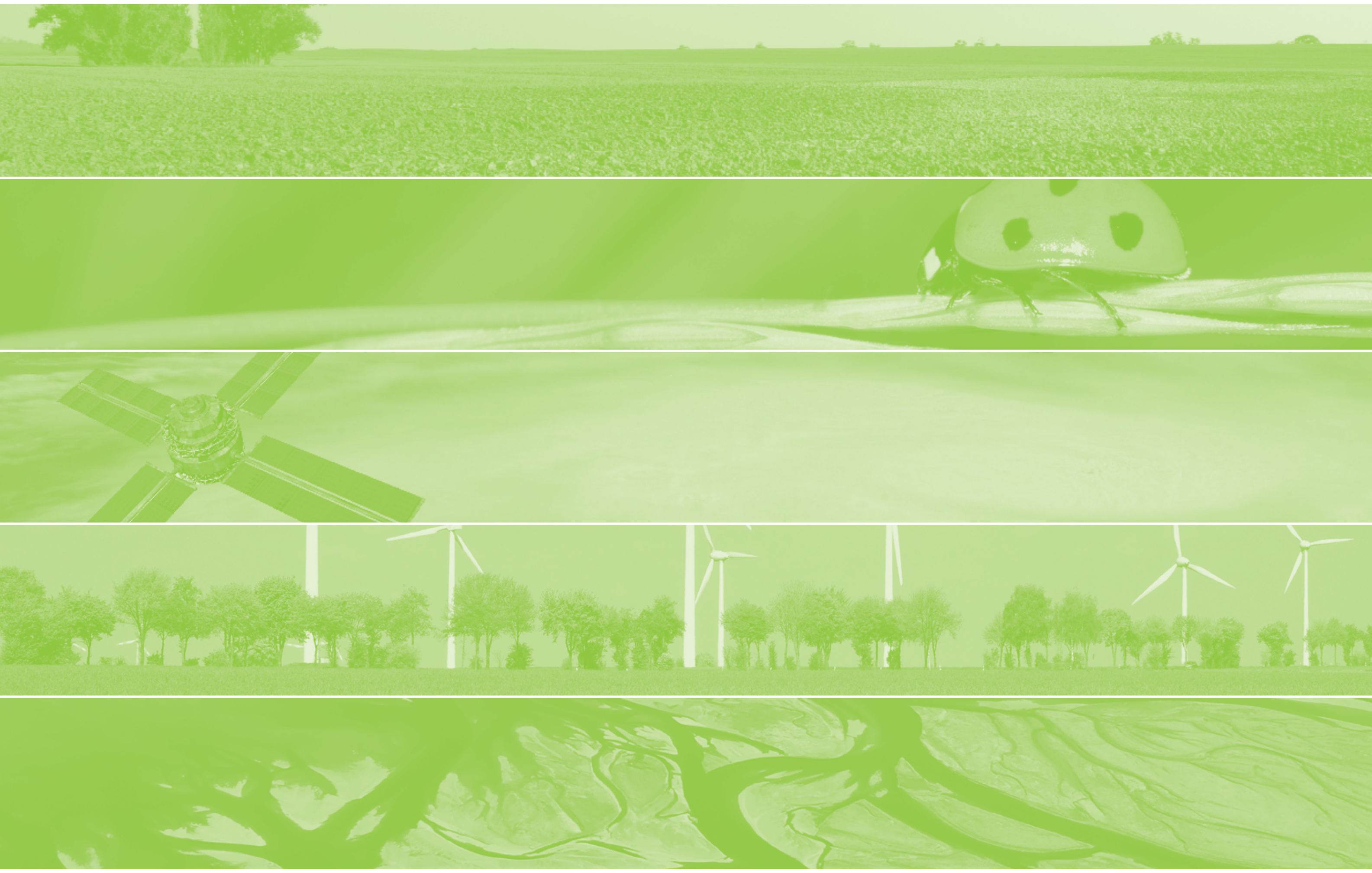
And for the special case of $p_2 = 0$:

$$t(x) = \frac{\varepsilon H_0}{R}\ln\left(\frac{L}{L-x}\,\frac{2B_0 + Lp_1}{2B_0 + p_1(L+x)}\right) \quad \text{[A7.6]}$$

Appendix 8 Area distribution of hydrogeological classes

Table A8.1
Agricultural area of the hydrogeological typology which was used for the selection of field study locations

Hydrogeo class	Netherlands		Agriculture in the NL		
	ha	% of NL	ha	% of HGclass	% of agriculture
a	84,383	2.4	53,337	63.2	2.5
b	1,169,979	33.5	616,112	52.7	29.1
c	49,303	1.4	34,792	70.6	1.6
d	448,947	12.8	272,638	60.7	12.9
e	564,577	16.1	332,868	59.0	15.7
f	1,179,463	33.7	810,900	68.8	38.2
NL	3,496,652	100.0	2,120,646	60.6	100.0



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