Rewetting polder Krimpenerwaard for nature conservation

Trends and patterns in meadow bird diversity, water quality and vegetation composition

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Photo front page: Overview of Berkenwoude, part of polder Krimpenerwaard. Photo: Suzanne Kanters.

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LAYMAN SAMENVATTING

Het Nederlandse landschap staat bekend om zijn lange geschiedenis in landbouw. Sinds de industriële revolutie is de landbouw steeds intensiever geworden. De intensifiëring van de landbouw heeft geleid tot een sterke afname in biodiversiteit. Vogelsoorten die in agrarisch gebied kunnen voorkomen zijn hier een goed voorbeeld van. Tussen 1990 en 2015 zijn bijna alle soorten vogels die in boerenland kunnen voorkomen in aantallen afgenomen. Van sommige vogelsoorten, bijvoorbeeld de veldleeuwerik, grutto, kluut en kemphaan, zijn de aantallen met meer dan 50% afgenomen. Om de afname in biodiversiteit in het landelijke gebied te stoppen worden door landelijke en provinciale overheden verschillende maatregelen genomen. Een van de maatregelen is het verhogen van het waterpeil, waardoor delen van graslanden gedurende een deel van het jaar onder water komen te staan. Deze situatie lijkt sterk op de situatie van vroeger, vóórdat graslanden drooggemalen werden ten behoeve van de landbouw.

In polder Berkenwoude en De Nesse in de Krimpenerwaard (Zuid-Holland) zijn op initiatief van de Provincie Zuid-Holland en met medewerking van het Zuid-Hollands Landschap in 2017 slootoevers afgevlakt en is het waterpeil verhoogd met als doel een toename in biodiversiteit te bewerkstelligen. Hoewel vogeldiversiteit, waterkwaliteit en vegetatiesamenstelling al sinds lange tijd wordt gemonitord in de Krimpenerwaard, was nog niet onderzocht hoe bovenstaande variabelen hebben gereageerd op de verhoging van het waterpeil. Daarom is onderzocht of en hoe weidevogeldiversiteit, waterkwaliteit in termen van nutriëntconcentraties en doorzicht en vegetatiecompositie zijn veranderd sinds de verhoging van het waterpeil. Hiervoor zijn bestaande gegevens van vogelterritoria (Zuid-Hollands Landschap), waterkwaliteit (Hoogheemraadschap Schieland en de Krimpenerwaard) en vegetatiesamenstelling (Provincie Zuid Holland) van een periode ten minste tien jaar geanalyseerd. Aansluitend is de literatuur geraadpleegd om de gevonden patronen en trends in een perspectief te plaatsen.

De weidevogeldiversiteit is na de waterpeilverhoging toegenomen in Berkenwoude en De Nesse. Deze toename verklaard kan worden door een verbetering van de habitat voor weidevogels en door een toename in voedselbeschikbaarheid.

Waterkwaliteit in termen van nutriëntbeschikbaarheid en doorzicht van het slootwater varieerde aanzienlijk in Berkenwoude en De Nesse. Effecten van de waterpeilverhoging op de waterkwaliteit in sloten waren niet eenduidig. Nutriëntbeschikbaarheid en doorzicht verschilden in Berkenwoude niet significant voor en na de waterpeilverhoging. In De Nesse nam zowel de fosfaatconcentratie en het doorzicht af na de verhoging van het waterpeil. Stikstofconcentraties verschilden niet. Binding van fosfaat aan ijzer zou theoretisch het mechanisme achter de afname van fosfaat kunnen zijn, omdat dit leidt tot het neerslaan van fosfaat. Dit verdient echter nader onderzoek.

Waterpeilverhoging leidt over het algemeen tot een verandering in de vegetatiesamenstelling waarbij ook de diversiteit van de vegetatie toeneemt. In Berkenwoude en De Nesse heeft de peilverhoging geresulteerd in vegetatietypes en dominante soorten die gekenmerkt worden door een ietwat hoger vochtgehalte in de bodem. De soortenrijkdom veranderde niet. In de regel leidt waterpeilverhoging tot een afname in bedekkingsgraad van aquatische vegetatie als gevolg van de ophoping van het toxische sulfide en door interne eutrofiëring. In Berkenwoude en De Nesse nam de vegetatiebedekking van ondergedoken waterplanten ook af in de proefvlakken.

De algemene conclusie van dit onderzoek luidt dat waterpeilverhoging een belangrijke rol kan spelen bij het herstellen van vogelpopulaties van open graslandschappen. Voorzichtig toonden zich ook al enkele effecten van de vernatting op de ontwikkeling van moerasvegetaties. De vernatting leidde niet overal tot een verhoogde nutriëntenbelasting in de sloten. Meer onderzoek over een langer tijdsframe en met meer proefvlakken in de steekproef is nodig voor de effecten van vernatting op de waterkwaliteit en vegetatiecompositie. Voornamelijk voor de vegetatie kan het enkele jaren duren voordat de effecten zichtbaar worden, waardoor monitoring gedurende langere tijd na een peilverhoging nodig is.

ABSTRACT

Landscapes in the Netherlands are characterised by a long agricultural history. The intensification of these agricultural practices since the industrial revolution has resulted in a strong decline in biodiversity across several species groups, including birds and plants. Restoring water levels to a level that enable small-scale periodic flooding is one of the measures that should help stopping the biodiversity decline. Creating a landscape in which grasslands are periodically flooded is comparable to the situation that existed before agricultural land was drained in order to execute intensive agriculture.

Berkenwoude and De Nesse are two parts of polder Krimpenerwaard (the Netherlands), which is a former area of intensive agriculture. On the initiative of the Province of South Holland and in collaboration with Zuid-Hollands Landschap, water levels have been increased in 2017 in Berkenwoude and De Nesse to increase biodiversity. However, the effects of this water level increase have not been investigated yet, even though bird diversity, water quality and vegetation composition has been monitored over time. Therefore, this study aimed to explore how this water level restoration affected bird biodiversity, water quality and vegetation composition one year after rewetting.

In Berkenwoude and De Nesse, bird diversity increased after the water level increase. The literature study revealed that as a result of the water level increase, the landscape may have become more favourable for birds and food availability may have increased.

Water quality in terms of nutrient concentrations and water transparency showed large variation in Berkenwoude and De Nesse. No uniform effect of the water level increase on water quality was present. In Berkenwoude, no differences in water transparency and nutrient concentrations existed before and after the water level increase. In De Nesse, both phosphate concentrations and transparency decreased after the water level increase. Total nitrogen levels remained stable. Precipitation of phosphate due to phosphate binding to iron is the underlying mechanism that may explain the decrease in phosphate levels.

The performed literature study showed that rewetting does not only leads to a terrestrial vegetation change towards species that prefer a higher moisture content, it can result in a more diverse vegetation composition as well. Rewetting led to a slight difference in terrestrial vegetation type and dominant species in Berkenwoude and De Nesse towards types and species that are associated with slightly higher soil wetness. Species richness remained similar. Regarding aquatic vegetation, the literature research showed that rewetting generally results in a decrease in vegetation cover due to the accumulation toxic sulphides and increased eutrophic conditions. This decrease in vegetation cover was detected in Berkenwoude and De Nesse as well.

Overall, the conclusion that rewetting helps to increase bird biodiversity was drawn. Moreover, some effects of rewetting in Krimpenerwaard on water quality and vegetation composition already became apparent. These effects must be further investigated in the future as it generally takes several years, especially for vegetation composition, for effects of rewetting to become apparent. Therefore, I recommend to elaborate the current study in time and in space.

TABLE OF CONTENTS

1. INTRODUCTION

1.1 Context

Intensive agricultural activity is one of the major drivers of the worldwide decline in biodiversity (Donald, Green, & Heath, 2001; Kleijn et al., 2012). Whereas extensive agriculture is vital for nature conservation, high-intensity farming has a highly detrimental effect on species richness and abundance (Reidsma, Tekelenburg, Van Den Berg, & Alkemade, 2006). Biodiversity is affected by intensive agriculture on both local and landscape scales. On a local scale, fertiliser and pesticide use, as well as crop rotation and grazing intensity affect biodiversity (Kleijn et al., 2012). At landscape scales, hydrological changes, habitat fragmentation and atmospheric deposition are problems related to biodiversity decreases (Kleijn et al., 2012). Intensive agriculture leads to a homogenisation of the landscape, both on local scales and on landscape scales (Flohre et al., 2011; Luoto, Rekolainen, Aakkula, & Pykälä, 2003). In Europe, increasing uniformity of habitats and their spatial arrangement is the prevailing trend in landscape development (Ihse, 1995; Luoto et al., 2003; Morris, 2000).

Another problem that is related to intensive agriculture is that areas that are less suitable for intensive agriculture are abandoned (Ernst, Tscharntke, & Batáry, 2017; Plieninger et al., 2015; Queiroz, Beilin, Folke, & Lindborg, 2014). Even though land abandonment can be beneficial for biodiversity if the abandoned areas are managed in a proper way (Foster, Kindscher, Houseman, & Murphy, 2009; Queiroz et al., 2014; Yamanaka, Akasaka, Yabuhara, & Nakamura, 2017), the majority of former agricultural lands are subjected to vegetation succession, leading to a loss of open areas that were former habitats of insect, bird and plant species preferring open habitats (Ernst et al., 2017; Queiroz et al., 2014; Uchida & Ushimaru, 2014).

Landscapes in the Netherlands are characterised by agriculture throughout a long history (Dekkers, 2002). More than 7000 years ago, the nomadic existence was replaced by agricultural settlements (Dekkers, 2002). These were small-scale settlements with extensive agriculture. In the late middle ages, the landscape radically changed as a result of water levels being governed by drainage (Vermaat, Goosen, & Omtzigt, 2007). This increased agricultural opportunities and led to an increase in agricultural activities in the centuries hereafter (Feng, 1998; Vermaat et al., 2007). Moreover, the start of the industrial revolution at the end of the 19th century marks another change in agriculture. Industrial developments augmented the demand for agricultural products and resulted in improved infrastructure that was essential for agricultural development (De Jong & Van Zanden, 2014; Feng, 1998). Around 1880, the first chemical fertilisers came into use, which strongly increased yields (Feng, 1998; Knibbe, 2000). After the end of World War II, a second modernisation phase took place. Agricultural activities changed again due to innovation, mechanisation intensification and area upscaling, leading to a more than 5-fold increase in production between 1950 and 2015 (CBS, 2017).

As a consequence of the intensification of agriculture in the Netherlands, a decline in biodiversity is observed across several species groups, including meadow birds and plants (Bobbink, Hornung, & Roelofs, 1998; Roland Bobbink & Willems, 1993; CBS, 2015; Pannek, Duprè, Gowing, Stevens, & Diekmann, 2015; Van Turnhout, Foppen, Leuven, Siepel, & Esselink, 2007). Between 1990 and 2015, the Living Planet Index (LPI) for almost all bird species that are characteristic for moist agricultural meadows declined in the Netherlands (figure 1.1) (CBS, 2015).

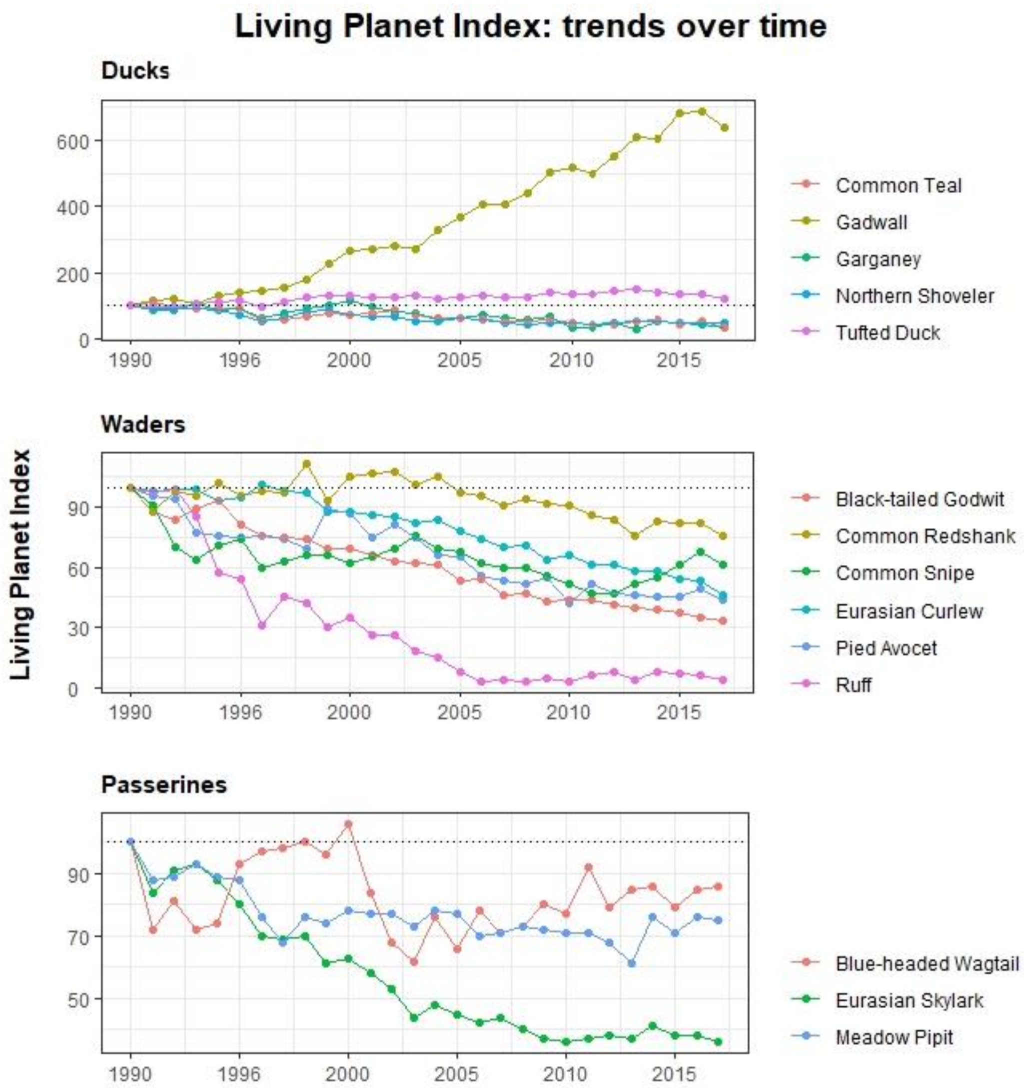

Figure 1.1 Living planet index (LPI) for the characteristic Dutch meadow bird species (according to the Dutch nature type N13.01: moist meadow bird grassland (Bij12, n.d.)). The number of individuals of the species in 1990 are the reference values of the LPI (LPI = 100). LPI values in the following years indicate the relative abundances compared to 1990. Adapted from: CBS, 2015.

To bring this declining trend to a halt, several measures have been taken. These include regional and national projects (e.g. Provincie Groningen, 2014; Westerink, Plomp, Ottburg, Zanen, & Schrijver, 2018), but also European Union-broad action plans (e.g. European Commission, 2019; Primdahl, Peco, Schramek, Andersen, & Oñate, 2003; Verhulst, Kleijn, & Berendse, 2007). Even though the programs have varying effectiveness in biodiversity conservation and restoration, the national and international measures all aim to increase biodiversity by restoring (semi-)natural conditions (Kleijn et al., 2004; Verhulst et al., 2007; Westerink et al., 2018).

The Dutch polder Krimpenerwaard, situated in the west of the Netherlands, is characterised by grasslands with ditches in between. The grasslands have a long agricultural history (Bijlmer & Stempher, 2018). Before the 1990s, land use aimed at improving the agricultural status of the area (Bijlmer & Stempher, 2018). However, starting from the 1990s, nature conservation became increasingly more important in Krimpenerwaard, as people started to realise that the decrease in floral and faunal diversity would continue if no measures were taken (Van Veen, Ten Brink, Braat & Melman, 2008; Wereld Natuur Fonds, 2015). Currently, a substantial part of Krimpenerwaard is nature reserve.

These nature reserves are under tenure by farmers. The farmers have to manage the land according to agreements with Zuid-Hollands Landschap (ZHL), which is a non-governmental Dutch nature conservation organisation. Benefits of this management regime that include farmers are that management costs for ZHL are minimal, while local farmers have some additional income as they are allowed to have some cattle or to sell mowed hay.

Water levels have been increased in 2017 in part of the nature reserve areas of Krimpenerwaard on the initiative of the Province of South Holland. Water levels were increased as a measure to increase biodiversity, since goals for biodiversity conservation and restoration were not met. As polder Krimpenerwaard exists of grasslands with ditches in between, the water level increase in the ditches leads to small-scale and temporal flooding of the grasslands. It is hypothesised that the water level increase causes the area to become more similar to the historic situation, when vegetation was much more diverse and meadow birds flourished. The Province of South Holland aimed to increase meadow bird abundance and increase vegetation diversity. Bird species that were aimed to be attracted are species that are characteristic for the nature type *moist meadow bird grasslands* (Dutch nature type N13.01). This nature type includes Black-tailed godwit (*Limosa limosa*), Common redshank (*Tringa totanus*), Common snipe (*Gallinago gallinago*), Eurasian curlew (*Numenius arquata*), Pied avocet (*Recurvirostra avosetta*), Ruff (*Philomachus pugnax*), Northern shoveler (*Anas clypeata*), Garganey (*Anas querquedula*), Eurasian teal (*Anas crecca*), Gadwall (*Mareca strepera*), Tufted duck (*Aythya fuligula*), Eurasian skylark (*Alauda arvensis*), Meadow pipit (*Anthus pratensis*), Western yellow wagtail (*Motacilla flava*). For a landscape to be recognised as a *moist meadow bird grasslands*, at least 35 territories of the above stated bird species must be present per 100 hectare (BIJ12, n.d.). Previous studies indicate that restoring water levels helps to attract these species (Belting, 2007; Blüml, 2012; Görn, Schulze, & Fischer, 2015; Mischenko, Sukhanova, & Zöckler, 2014). However, the effects of the water level increase on meadow birds in Krimpenerwaard have not been studied yet, even though data is available.

Before the modernisation of agriculture in Krimpenerwaard, the area was characterised by a large area of fen meadow, which exceeded 10.000 hectares (De Vries, 1929). However, as a consequence of nutrient enrichment due to fertiliser use, fen meadow area decreased to 70-80 hectares in 1924 and only 1.5 hectares was left in 2010 (De Vries, 1929; Verhoeven, Barendregt, & Van De Riet, 2010). Together with periodic topsoil stripping, increasing water levels can lead to vegetation changes towards a more diverse, less eutrophic vegetation cover. Even though fen meadows are not likely to have restored yet due to a nutrient load that is still too high to support fen meadow plant species, alterations in vegetation cover are likely to have occurred and to continue changing as a result of the water level increase. However, any patterns in the development of the vegetation before and after the water level increase remain unknown.

The increase in water levels may influence water quality in terms of nutrient availability and water transparency as well. Due to an increase in water levels, phosphate from the sediment and the soil can be mobilised, leading to temporary eutrophication, both on land and in ditches (Bobbink, Hart, Van kempen, Smolders, & Roelofs, 2007; Lamers, Lucassen, Smolders, & Roelofs, 2005; Zak & Gelbrecht, 2007). Moreover, soil nitrates runoff to ground- and surface waters (Bobbink et al., 2007; Lamers et al., 2005; Zak & Gelbrecht, 2007). Therefore, a peak in nutrient availability is likely to be observed shortly after the water level increase, accompanied by a decrease in water transparency due to algal blooms. For Krimpenerwaard, long time series with water quality data (nutrient concentrations, water transparency) measured by the regional water authority Schieland en de Krimpenerwaard (HHSK) exist. Nevertheless, this data has not been evaluated yet in terms of the possible effects of the water level increase on water quality.

Monitoring of water quality, meadow bird territories and vegetation composition is regularly performed in Krimpenerwaard (Terlouw, 2015; Van der Winden et al., 2018; Van Donk, Courbois, Koenders, & Van der Winden, 2019). Nonetheless, the long-term trends in meadow bird diversity and vegetation composition and the effects of the increase in water level on meadow bird territories and vegetation composition throughout part of the polder have not been evaluated yet. As a result, a knowledge gap exists regarding the effectiveness of water level increase on meadow birds, vegetation diversity and water quality. Therefore, the aim of this study is to gain insight in the relation between the water level increase and water quality, meadow bird diversity and vegetation composition. This insight is gained by investigating the available Krimpenerwaard data. Subsequently, mechanisms explaining the observed trends are discussed and comparisons with other areas in which water levels have been increased are examined by performing a literature study on the effects of water level increases in meadows and abandoned agricultural lands that became nature reserves. By gaining insight in the relations between increasing water levels and the underlying mechanisms, this research contributes to a better understanding of measures that can be taken to effectively halt the trend in biodiversity decline in the Netherlands. As ZHL aims to improve vegetation and meadow bird diversity within other nature conservation areas in Krimpenerwaard as well, this research contributes to the implementation of measures in other areas in Krimpenerwaard. Moreover, this research helps to gain insight in the temporary scales at which the effects of measures that are taken to improve biodiversity and water quality can be detected. It should be noted that this research focuses on investigating correlations between a water level increase and water quality, meadow bird diversity and vegetation composition rather than investigating causal relations.

1.2 Research questions

This research focuses on examining the relations between a water level increase and meadow bird diversity, water quality and vegetation composition. Therefore, the most important question that is answered is: "*How does a water level increase in a former agricultural area affects meadow bird biodiversity, water quality and vegetation composition*?".

To answer this question, three sub-questions are formulated:

Meadow birds
How does an increase in water level affects the biodiversity of meadow birds in terms of species richness and abundance?

Water quality
To what extent does an increase in water level affect ditch water quality in terms of nutrient concentrations and water transparency?

Vegetation types
How does an increase in water level affect vegetation composition?

1.3 Hypotheses

Birds
Meadow birds prefer wet, open grasslands as a habitat (Besnard et al., 2013; Milsom et al., 2000; Zmihorski, Pärt, Gustafson, & Berg, 2016). The worldwide decline in meadow bird species numbers has been linked to drainage for agriculture with consequent loss of food availability (Bradbury & Kirby, 2006; Fraixedas et al., 2017; Stanton, Morrissey, & Clark, 2018; Van der Weijden & Guldemond, 2006). As a result, I hypothesise that that 1) increased water levels in polder Krimpenerwaard have led to an

increase in meadow birds compared to the former low water level situation and 2) an increase in food availability is the mechanism underlying the increase in meadow bird abundance.

Water quality

Nature reserves that are former agricultural areas are eutrophic. Soil nitrates that are inherited from the agricultural history generally run off to the surface water or ground water (Lamers, Lucassen, Smolders, & Roelofs, 2005). Meanwhile, under dry conditions, phosphorus remains bound to the soil (Lamers, Tomassen, & Roelofs, 1998). When water levels are increased, part of the bound phosphorus that is stored in the soil is mobilised, leading to a decrease in water quality due to internal eutrophication (Lamers et al., 2005; Olde Venterink, Davidsson, Kiehl, & Leonardson, 2002). The eutrophic water conditions can lead to a system shift from a clear, macrophyte-dominated state to a turbid, algae-dominated state (Scheffer, Hosper, Meijer, Moss, & Jeppesen, 1993). As the process of reducing the nutrient conditions can last decades to centuries (Lamers et al., 2005), it is hypothesised that 1) the increase in water level leads to a significant increase in nutrient concentrations and 2) this increase in nutrient concentrations leads to a significant decrease in water transparency.

Vegetation types

Large-scale drainage has led to peat decomposition, soil acidification and iron depletion (Van Andel & Aronson, 2006). This, together with intensive agrochemical use, has led to a decline in grassland diversity throughout the Netherlands, where generalist, eutrophic species are the only remaining species (Blomqvist, Vos, Klinkhamer, & Ter Keurs, 2003; De Snoo, Naus, Verhulst, Van Ruijven, & Schaffers, 2012). However, rewetting the soil has been found not to automatically lead to recovery of the former vegetation (Van Andel & Aronson, 2006). Due to the system having reached an alternative stable state, return of the original vegetation types is difficult to achieve, even if nutrient loads are reduced to former levels as well (Scheffer et al., 1993; Van Andel & Aronson, 2006). Nevertheless, soil rewetting may result in the (re-)establishment of species preferring high water levels (Casanova & Brock, 2000). Therefore, it is hypothesised that 1) no significant differences exist in vegetation composition and -diversity before and after water level increase in polder Krimpenerwaard and 2) species richness increases due to the (re-)establishment of water-preferring species as a result of the increased water levels in Krimpenerwaard.

2. METHODS

2.1 Study site

The Dutch polder Krimpenerwaard is situated in the west of the Netherlands, between the cities of The Hague, Rotterdam and Utrecht (figure 2.1). The total area size of Krimpenerwaard is approximately 170 km^2 (www.ontdekdekrimpenerwaard.nl). Two polders within Krimpenerwaard were chosen as focus area, as in these two polders, water levels were increased in June 2017. These areas include De Nesse and Berkenwoude (figure 2.1). Summer water levels were increased in June 2017, from -2.65m – -2.62m NAP (zero reference for elevation) to a level varying between -2.46m and -2.31m. Winter levels are currently even higher: ca. -2.10m NAP. In addition to the two polders De Nesse and Berkenwoude, a reference area was included for all analyses. In the reference area, water levels were not increased in June 2017. An area surrounding Berkenwoude and De Nesse was chosen as the reference area for the water quality analyses and vegetation analyses (figure 2.1). Polder Bilwijk was chosen as a reference for the bird data as most bird data was available for this polder (figure 2.1). Area size of De Nesse and Berkenwoude are ca. 300 ha and 165 ha, respectively. Bilwijk is approximately 130 ha, the reference area surrounding De Nesse and Berkenwoude is ca. 350 ha. Whereas De Nesse and the two reference areas are open polders where trees are largely absent, Berkenwoude is more closed, with presence of several tree stands including small forest patches.

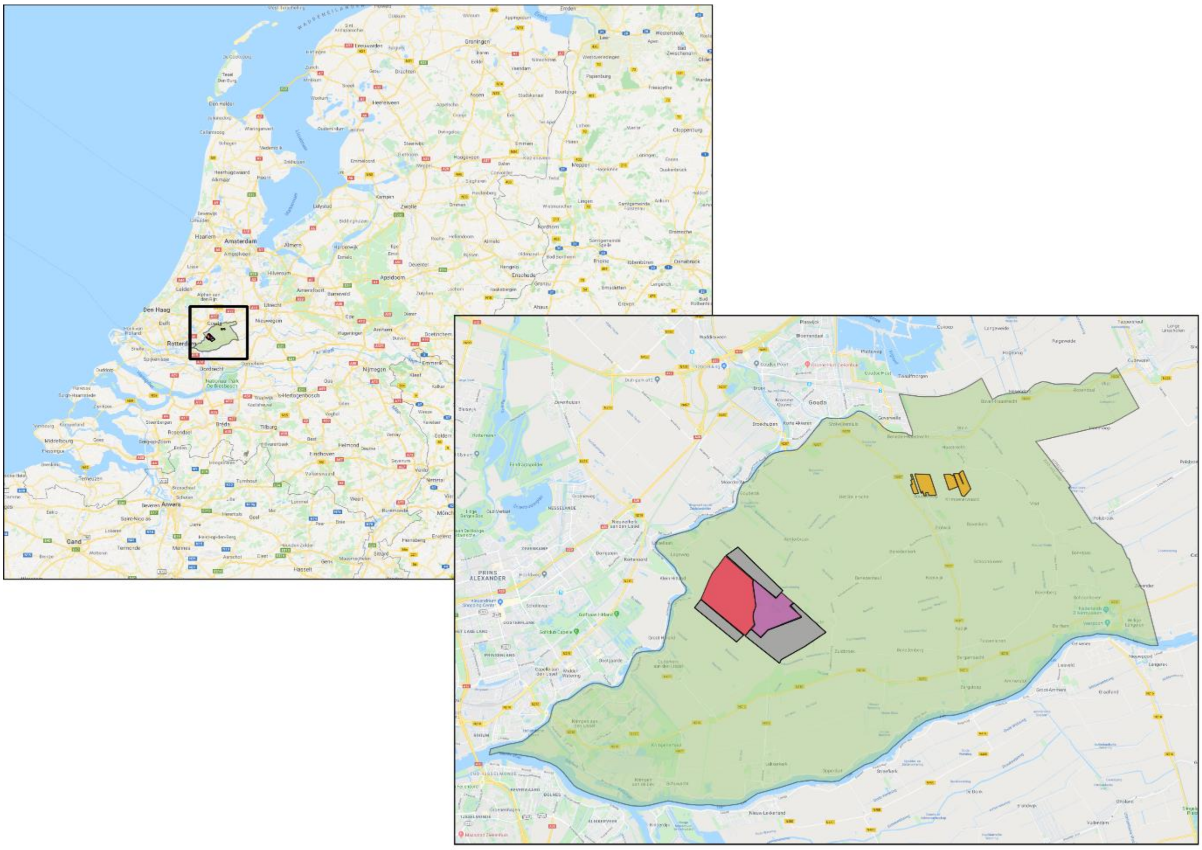

Figure 2.1 Location of Krimpenerwaard. The highlighted blocks are polder De Nesse (red), Berkenwoude (purple) and the reference areas. The reference area for the bird data analyses, Bilwijk, is highlighted in orange, the reference area for the water quality and vegetation composition analyses is highlighted in grey.

2.2 Bird data and analyses

In Bilwijk, bird territories were monitored in 2008, 2014 and 2018. In Berkenwoude, monitoring took place in 2011, 2014 and 2018 and in De Nesse, inventories of bird territories were made in 2011, 2014, 2015 and 2018. The data obtained during the monitoring events in 2008, 2011, 2014 and 2015 were retrieved from hard-copy reports and digitised in Microsoft Excel (2010) and in QGIS version 2.18.26 (QGIS Development Team, 2018). The monitoring outcomes of 2018 were already digitised in QGIS. The number of territories of the selected meadow bird species (Black-tailed godwit, Common redshank, Common snipe, Eurasian curlew, Northern shoveler, Garganey, Eurasian teal, Gadwall, Tufted duck, Eurasian skylark, Meadow pipit, Western yellow wagtail) were extracted from the QGIS data file and digitised in Microsoft Excel (2010). Ruff and Pied avocet were excluded from the analyses, as no individuals of these species were found in the investigated polders.

For all three polders – De Nesse, Berkenwoude, Bilwijk – a Living Planet Index was calculated. The first monitoring year of each of the polders was taken as the reference year (LPI = 100). To calculate the LPI for subsequent monitoring years, the geometric mean of the relative changes in species abundance was calculated. The presence of new species in the polders were indicated with a value of 100, while the disappearance of species in subsequent years was indicated with a value of 1. Abundances of newly arrived species were not taken into account: presence of a new species with an abundance of either 1 or 10 were both given a value of 100. The LPIs were analysed by means of R (R Core Team, 2018), using the packages "*ggplot2*" (Wickham, 2016) and *"reshape2"* (Wickham, 2007).

To examine whether significant differences in species abundance (per 100 ha) existed between polders and years, Kruskal-Wallis tests were used. Kruskal-Wallis tests were performed as the assumptions for ANOVA – random sample, equal variances, normal distribution - were not met.

2.3 Water quality data and analyses

Water quality has been measured in terms of nitrogen (N) concentrations, phosphate (PO_4^{3-}) concentrations and water transparency. In Krimpenerwaard, water quality has been monitored since the 1980s. Some sites were monitored over a long period, while others were only measured for a few years. Monitoring was performed once per month during the entire year. To be included in the water quality analyses, sites had to meet three criteria. First, the monitoring sites in De Nesse and Berkenwoude had to be within the area in which water levels had been increased. Hence, sample points at the border of the polders were excluded. Second, the monitoring period had to start before the water level increase in June 2017. Last, monitoring had to be continued to at least one year after the water level increase.

In Berkenwoude and De Nesse, four and three monitoring sites met these conditions, respectively (figure 2.2). In the reference area situated alongside Berkenwoude and De Nesse, three monitoring sites met the criteria (figure 2.2).

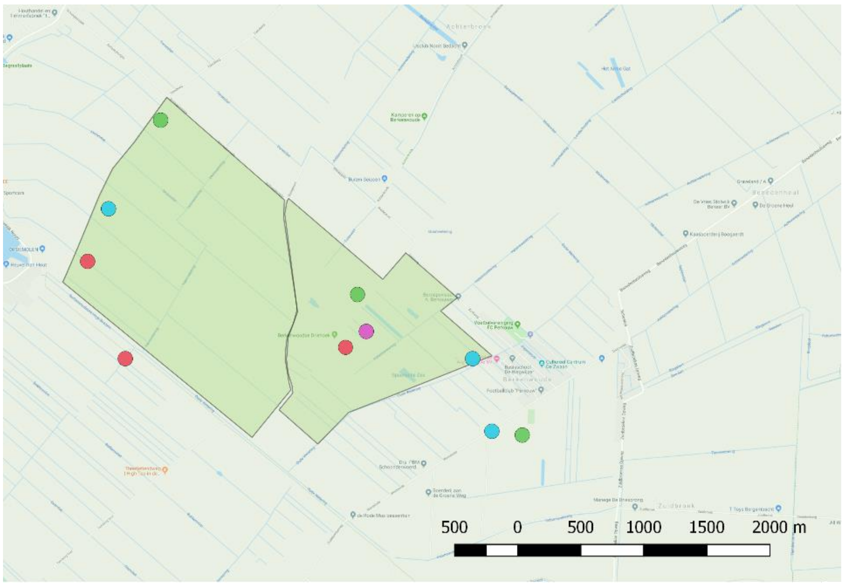

Figure 2.2 Monitoring sites in De Nesse, Berkenwoude and the reference area. Colours of the dots correspond with the numbers of the monitoring sites as given in the results section: red = site 1; green = site 2; blue = site 3; purple = site 4.

By means of R (R Core Team, 2018), graphs of the trends in total nitrogen concentrations, phosphate concentrations and water transparency that were measured in the polders were made. The packages *"ggplot2", "gridExtra"* and *"grid"* were used to visualise the data.

To investigate whether significant differences existed between the polders and between the monitoring sites within Berkenwoude, De Nesse and the reference area, Kruskal-Wallis tests were performed, as for all three parameters (nitrogen concentration, phosphate concentration and water transparency) ANOVA assumptions of the data being normal distributed and having equal variances were violated. Dunn's post-hoc tests (R package "*dunn.test*", Dinno 2017) were performed to find out which groups showed significant differences.

Moreover, to test whether significant differences in nitrogen and phosphate concentrations and water transparency existed before and after the water level increase, Wilcoxon's rank-sum tests were performed as well.

2.4 Vegetation data and analysis

Vegetation data including vegetation type, monitoring location, vegetation cover and species presence with abundance classes have been collected for the entire Krimpenerwaard. Vegetation monitoring started in the late 1970s and expanded during the 1990s. Vegetation cover per species was recorded using the Braun-Blanquet method. The Braun-Blanquet codes correspond to fixed cover percentages (table 2.1). Plot size of the monitored vegetation ranged between 20 m^2 and 100 m^2 for the terrestrial samples and between 105 m^2 and 400 m^2 for the aquatic samples. Vegetation was monitored between June and September.

To be included in the analyses, the monitoring locations had to meet three criteria. First, monitoring locations in De Nesse and Berkenwoude had to be situated within the area in which water levels had been increased. Second, the reference monitoring points where water levels were not increased in June 2017 had to be situated close to the polders Berkenwoude and De Nesse. Last, the monitoring period in De Nesse, Berkenwoude and the reference area had to range from at least January 2017 (6 months before the water level increase in De Nesse and Berkenwoude) to January 2018 (6 months after the water level increase in De Nesse and Berkenwoude).

Table 2.1 Conversion table of the Braun-Blanquet codes to cover percentages as used in the vegetation database from the Province of South Holland.

Cover code	Cover percentage	Cover code	Cover percentage
r	1	2b	18
+	2	3	38
1	3	4	63
2m	4	5	88
2a	8		

In Berkenwoude, eight different monitoring sites exist, of which five are situated at terrestrial sites and four in ditches covering aquatic vegetation (figure 2.3). In De Nesse, three different monitoring sites exist (figure 2.3). Terrestrial vegetation was monitored at site 1 and site 2, aquatic vegetation was monitored at site 1 and 3. (figure 2.3). Four reference monitoring sites met the criteria. Aquatic plants were monitored at all four sites, terrestrial plants were measured at two sites (figure 2.3).

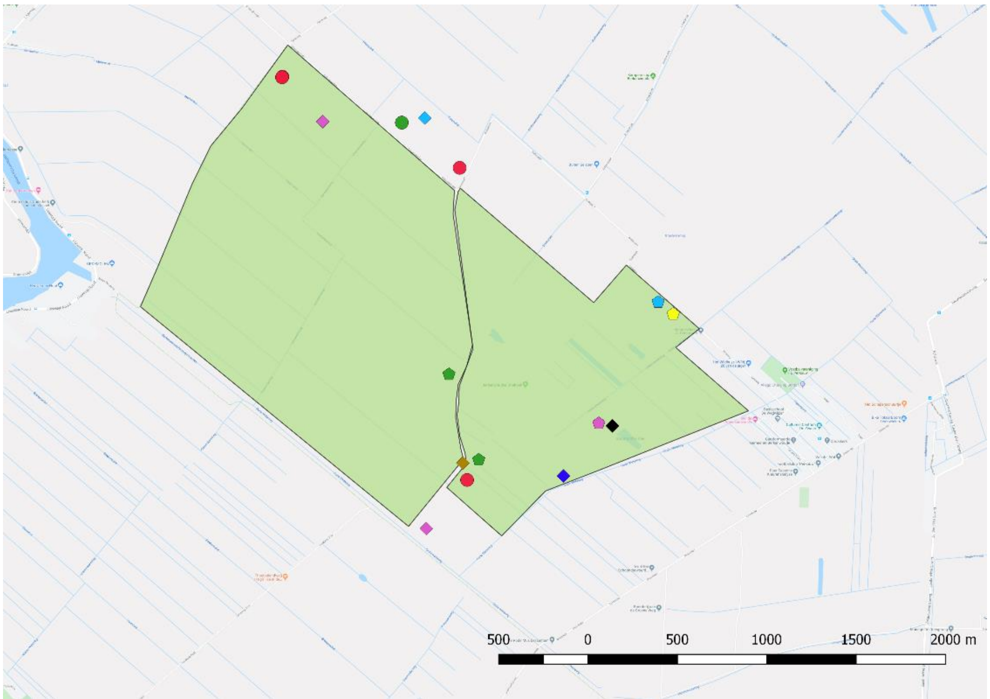

Figure 2.3 Locations of the monitoring sites in Berkenwoude (8), De Nesse (3) and the reference area (4). Circles indicate terrestrial and aquatic vegetation sample sites. Aquatic sample sites are indicated by diamonds. Pentagons signify terrestrial vegetation sample sites. Red = site 1; green = site 2; pink = site 3; light blue = site 4; yellow = site 5; brown = site 6; dark blue = site 7; black = site 8.

3. RESULTS

3.1 Meadow bird territories

Berkenwoude

Between 2011 and 2014, the total number of bird territories increased from 52 to 82 in Berkenwoude (table 3.1). Hereafter, the total number of territories decreased to a total of 73 in 2018, even though water levels have been increased in 2017. Even though the number of territories varied over time, no significant differences are found over the years ($X^2 = 1.30$, $p = 0.52$). Nevertheless, the LPI increased during this period (figure 3.1), indicating a diversity increase.

Densities of meadow bird territories per 100 ha ranged between 31 and 44 (table 3.1). In 2014 and 2018, more than 35 bird territories per 100 ha were present, indicating that the condition for moist meadow bird grasslands (number of territories per 100 ha ≥ 35, see introduction) was met during these years.

Table 3.1 Number of territories per meadow bird species in Berkenwoude (ca. 165 ha) over time. The number of territories per 100 ha is given between brackets. Species that were absent during all monitoring years (e.g. Eurasian curlew) are excluded from the table.

BERKENWOUDE			
Species	**2011**	**2014**	**2018**
Black-tailed godwit	21 (12.73)	29 (17.58)	21 (12.73)
Common redshank	6 (3.64)	6 (3.64)	11 (6.67)
Common snipe	0 (0)	1 (0.61)	0 (0)
Gadwall	15 (9.09)	15 (9.09)	31 (18.79)
Garganey	1 (0.61)	3 (1.82)	1 (0.61)
Meadow pipit	0 (0)	2 (1.21)	0 (0)
Northern shoveler	8 (4.85)	20 (12.12)	12 (7.27)
Tufted duck	1 (0.61)	5 (3.03)	7 (4.24)
Western yellow wagtail	0 (0)	1 (0.61)	0 (0.33)
TOTAL	52 (31.52)	82 (49.70)	73 (44.24)

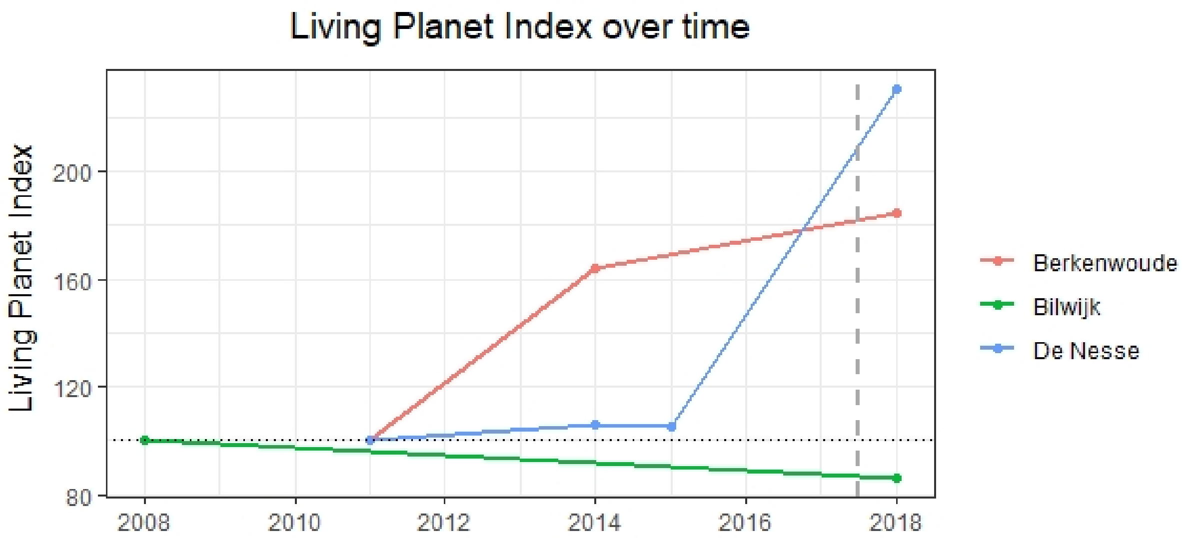

Figure 3.1 Living Planet Index of the selected meadow bird species (see methods section 2.2) in Berkenwoude, Bilwijk and De Nesse. In Berkenwoude and De Nesse, water levels have been increased in June 2017 (dashed line). In Bilwijk, water levels were not increased.

De Nesse

In De Nesse, the number of bird territories was much higher in 2018, after the water level increase, than in before (table 3.2). As a result, the LPI in De Nesse more than doubled: the LPI was 230 in 2018 compared to 100 (reference) in 2011 (figure 3.1).

The increase in the number of bird territories was most pronounced for the Black-tailed godwit, Common redshank, Gadwall, Northern shoveler and Tufted duck (table 3.2). Moreover, four species that were absent in 2011 and 2015 were present in 2018. Nevertheless, due to large variation in abundance per species (table 3.2), the increase in the number of bird territories is not significant (X^2 = 3.75, p = 0.29).

The condition that at least 35 meadow bird territories must be present in moist meadow bird grasslands was met after the water level increase, while before the water level increase, the condition was violated (table 3.2).

Table 3.2 Number of territories per meadow bird species in De Nesse (ca. 300 ha) over time. The number of territories per 100 ha is given between brackets. Species that were absent during all monitoring years (e.g. Eurasian teal) are excluded from the table.

DE NESSE				
Species	**2011**	**2014**	**2015**	**2018**
Black-tailed godwit	21 (7)	27 (9)	26 (8.67)	61 (20.33)
Common redshank	11 (3.67)	7 (2.33)	12 (4)	40 (13.33)
Common snipe	0 (0)	2 (0.67)	0 (0)	1 (0.33)
Eurasian curlew	0 (0)	1 (0.33)	0 (0)	0 (0)
Eurasian skylark	0 (0)	2 (0.67)	0 (0)	9 (3)
Gadwall	17 (5.67)	14 (4.67)	14 (4.67)	65 (21.67)
Garganey	2 (0.67)	3 (1)	3 (1)	7 (2.33)
Meadow pipit	0 (0)	5 (1.67)	6 (2)	2 (0.67)
Northern shoveler	7 (2.33)	16 (5.33)	9 (3)	31 (10.33)
Tufted duck	6 (2)	5 (1.67)	4 (1.33)	40 (13.33)
Western yellow wagtail	0 (0)	1 (0.33)	0 (0)	2 (0.67)
TOTAL	64 (21.33)	83 (27.67)	74 (24.67)	258 (86)

Bilwijk

In Bilwijk, water levels have not been increased in 2017. Nevertheless, the goals for moist meadow bird grasslands were met both in 2008 and in 2018 (table 3.3): in both years, the number of meadow bird territories per 100 ha exceeded 35 territories. Nevertheless, in 2018, the number of territories in Bilwijk had declined compared to 2008.

In addition to the number of territories, the LPI declined as well (figure 3.1). As for the other areas, the variation in the number of individuals per species was so large, that no significant difference in meadow bird abundance between 2008 and 2018 is found (X^2 = 0.77, p = 0.68).

Differences between the polders

Even though not significant, the three polders showed considerable variation in the number of territories in 2018. In De Nesse, 86 meadow bird territories per 100 ha were present in 2018, while this was much lower in Berkenwoude (44.24) and Bilwijk (42.31). Nevertheless, the relative species abundances were similar across the polders, with Black-tailed godwit and Gadwall being the most abundant species in all three polders. Common redshank and Northern shoveler had similar relative abundances in Berkenwoude, De Nesse and Bilwijk as well (table 3.1 – table 3.3). Nevertheless, Tufted duck had relatively high abundance in Berkenwoude and De Nesse, while its relative abundance in Bilwijk was much lower. The abundances of Common snipe, Eurasian skylark, Garganey, Meadow pipit and Western yellow wagtail were low, or the species were absent. In 2018, Eurasian curlew and Eurasian teal were absent in all three polders. Ruff and Pied avocet were absent in all monitoring years.

Table 3.3 Number of territories per meadow bird species in Bilwijk (ca. 130 ha) over time. The number of territories per 100 ha is given between brackets. Species that were absent during all monitoring years (e.g. Eurasian curlew) are not excluded from the table.

BILWIJK		
Species	**2008**	**2018**
Black-tailed godwit	37 (28.46)	21 (16.15)
Common redshank	14 (10.77)	9 (6.92)
Eurasian skylark	1 (0.77)	1 (0.77)
Gadwall	4 (3.07)	13 (10)
Meadow pipit	1 (0.77)	1 (0.77)
Northern shoveler	10 (7.69)	9 (6.92)
Tufted duck	3 (2.31)	1 (0.77)
TOTAL	70 (53.85)	55 (42.31)

3.2 Water quality

3.2.1 Nitrogen concentrations

Total nitrogen concentrations show large variation over time in the polders (figure 3.2): a minimum concentration of 1.1 mg L^{-1} was detected thrice in De Nesse (July 2010; May and June 2019) and once in the reference area (October 2013), while a maximum concentration of 13 mg L^{-1} was detected in Berkenwoude in July 2018 (appendix A figure 1). Nitrogen concentrations in Berkenwoude (4.56 ± 1.76 mg L^{-1}) are significantly higher than nitrogen concentrations in De Nesse (3.85 ± 1.48) and the reference area (4.05 ± 1.58) (appendix A table 1).

Berkenwoude

In Berkenwoude, nitrogen concentrations at site 4 (3.09 ± 0.90 mg L^{-1}) are significantly lower than nitrogen concentrations at site 2 (4.46 ± 1.26 mg L^{-1}) and site 3 (4.77 ± 2.46 mg L^{-1}), which are significantly lower than the concentrations at site 1 (5.17 ± 1.48 mg L^{-1}) (appendix A table 2).

Impacts of the water level increase on nitrogen concentrations are not detected in Berkenwoude (appendix A table 3). Nevertheless, after the water level increase, a peak in nitrogen concentration existed at two of the four monitoring sites in March 2018 (site 1) and July 2018 (site 3) (figure 3.2).

De Nesse

In De Nesse, the nitrogen concentration at site 1 (4.01 ± 1.64 mg L^{-1}) and site 3 (4.11 ± 1.28 mg L^{-1}) are significantly higher than the nitrogen concentration at site 2 (3.42 ± 1.21 mg L^{-1}) (appendix A table 4).

The water level increase in June 2017 resulted in a significant decrease in total nitrogen concentration at site 1 from 4.20 ± 1.66 mg L^{-1} to 3.05 ± 1.13 mg L^{-1} and at site 3 from 4.59 ± 1.38 mg L^{-1} to 3.55 ± 0.90 mg L^{-1} (appendix A table 3). At site 2, the water level increase had no significant effect on the total nitrogen concentration (appendix A table 3). Whereas in Berkenwoude, peaks in nitrogen concentration are observed at two sites, peaks are absent in De Nesse (figure 3.2).

Reference area

The nitrogen concentration at site 2 (4.55 ± 1.76 mg L^{-1}) is significantly higher than the nitrogen concentration at site 1 (3.77 ± 1.38 mg L^{-1}) and site 3 (3.75 ± 1.42 mg L^{-1}) (appendix A table 5).

Differences in nitrogen concentration before June 2017 and after June 2017, when water levels have been increased in Berkenwoude and De Nesse, do not exist in the reference area (appendix A table 3).

General patterns in nitrogen concentration

In general, the variation in nitrogen concentrations is large in polder Krimpenerwaard, both within and between the three different locations. Any effects of the water level increase can only be detected in De Nesse. In the reference area, where water levels have not been restored in June 2017, per site variation in nitrogen concentrations over time is not significant, implying that other factors affecting nitrogen concentrations in Krimpenerwaard are absent.

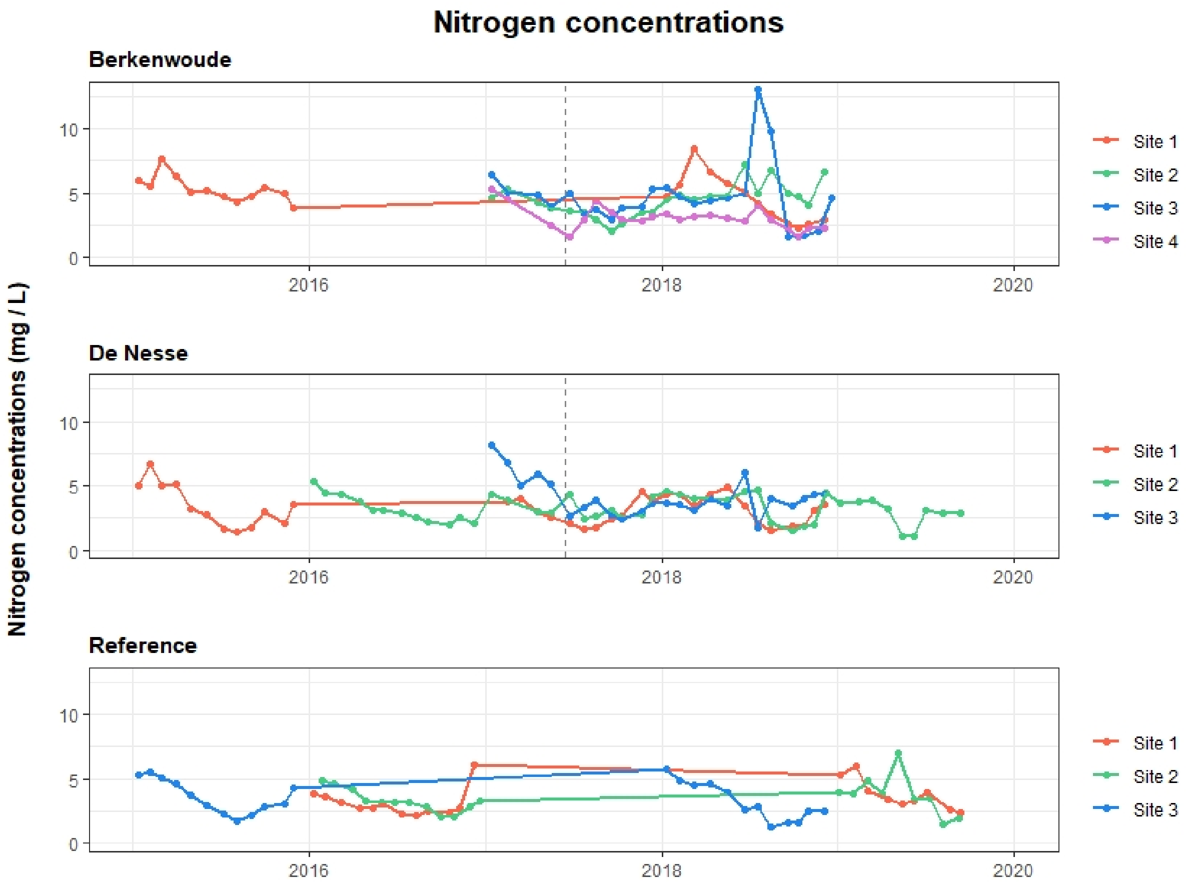

Figure 3.2 Total nitrogen concentrations (mg L^{-1}) in Berkenwoude, De Nesse and the reference area. The dashed line indicates the water level increase in Berkenwoude and De Nesse. In the reference area, water levels have not been increased.

3.2.2 Phosphate concentrations

Phosphate concentrations show large variations between the polders. Phosphate concentrations are highest in the reference area (0.75 ± 0.44 mg L^{-1}). In De Nesse (0.56 ± 0.46) and Berkenwoude (0.29 ± 0.35), phosphate concentrations are significantly lower than in the reference area (appendix A table 6). Phosphate concentrations in Berkenwoude are significantly lower than phosphate concentrations in De Nesse as well (appendix A table 6). Despite the differences in phosphate concentration between the locations, coinciding peaks in phosphate concentration existed during the late spring of 2018 in Berkenwoude, De Nesse and the reference area (figure 3.3). In addition, parallel peaks existed in De Nesse and the reference area in June and August 2019 (figure 3.3).

Berkenwoude

Despite the large variation in phosphate concentrations at the different monitoring sites in Berkenwoude, phosphate concentration at site 1 (0.44 ± 0.41 mg L^{-1}) and site 3 (0.36 ± 0.31 mg L^{-1}) are significantly higher than the phosphate concentrations at site 2 (0.04 ± 0.04 mg L^{-1}) and site 4 (0.14 ± 0.15 mg L^{-1}) (appendix A table 7). Moreover, the phosphate concentration at site 4 was significantly higher than the phosphate concentration at site 2 (appendix A table 7).

At site 1 and site 3, peaks in phosphate concentration after the water level increase exist (figure 3.3). At site 1, this peak concentration was measured in May 2018 and coincides with peaks observed in De Nesse and the reference area (figure 3.3). At site 3, two peaks exist: the first peak was between December 2017 and January 2018, the second peak coincided with the peak at site 1 and peaks in De Nesse and the references area (April - May 2018, figure 3.3). Significant differences in phosphate concentration before and after the water level increase do not exist in Berkenwoude (appendix A table 8).

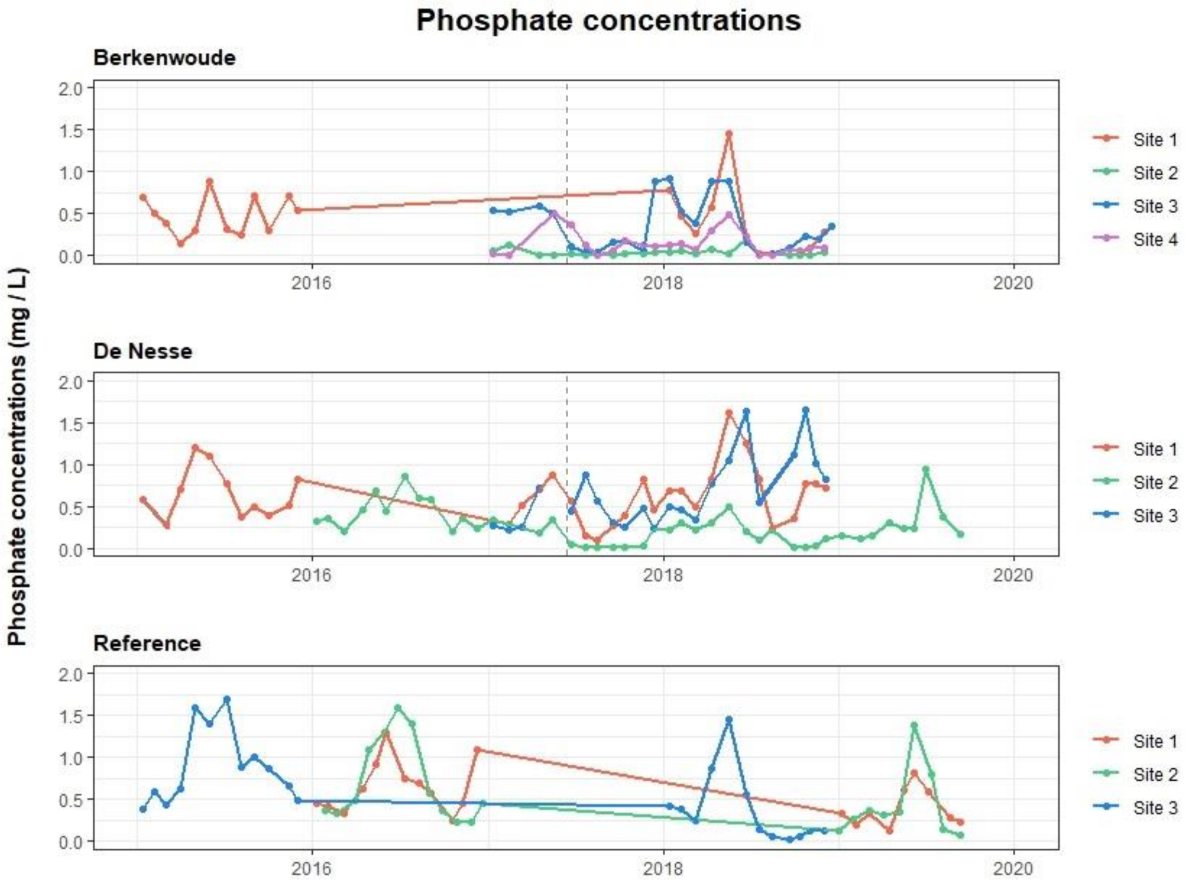

Figure 3.3 Phosphate concentrations (mg L^{-1}) in Berkenwoude, De Nesse and the reference area. The dashed line indicates the water level increase in Berkenwoude and De Nesse. In the reference area, water levels have not been increased.

De Nesse

The phosphate concentration at site 2 in De Nesse is significantly lower (0.27 ± 0.22 mg L^{-1}) than at site 1 (0.71 ± 0.48 mg L^{-1}) and site 3 (0.64 ± 0.50 mg L^{-1}; appendix A table 9).

Peaks in phosphate concentrations after the water level increase are observed at all three sites (figure 3.3). At site 1, the phosphate concentration peaked in May 2018. The peak at site 2 was less intense and later: phosphate levels were maximal in July 2019 (figure 3.3). At site 3, phosphate concentrations peaked for the first time in June 2018, followed by a second peak in October 2018 (figure 3.3). The peak at site 1 and the first peak at site 3 coincide with peaks that are present in Berkenwoude and the reference area (figure 3.3). Despite the observed peaks, the relation between the increase in water level and phosphate concentration is not uniform. The increased water levels did not have a significant effect on the phosphate concentration at site 1 (appendix A table 8). However, after the water level increase, phosphate concentrations were significantly lower (0.19 ± 0.20 mg L^{-1}) at site 2 than before the water level increase (0.32 ± 0.19 mg L^{-1}). In contrast, at site 3, the phosphate concentration was significantly higher after the water level increase (0.80 ± 0.52 mg L^{-1}) than before the increase (0.50 ± 0.44 mg L^{-1}).

Reference area

In the reference area, phosphate levels are significantly lower at site 2 (0.64 ± 0.42 mg L^{-1}) than at site 1 (0.81 ± 0.47 mg L^{-1}) and site 3 (0.82 ± 0.42 mg L^{-1}) (appendix A table 10). Phosphate concentration peaked in May 2018 (site 3) and June 2019 (site 1 and 2) (figure 3.3).

Even though water levels did not increase in the reference area, the phosphate concentration at site 1 and site 3 was significantly lower after June 2017 than before this period (appendix A table 8; appendix A figure 2). At site 1 and 3, phosphate concentrations before June 2017 were 0.85 ± 0.46 mg L^{-1} and 0.89 ± 0.38 mg L^{-1}, respectively. After June 2017, phosphate concentrations were 0.39 ± 0.23 mg L^{-1} and 0.37 ± 0.42 mg L^{-1}, respectively. Despite the decrease in phosphate concentration at site 1 and 3, peaks that coincide with peaks in the other polders were present after June 2017. No significant differences over time exist at site 2.

General patterns in phosphate concentration

Phosphate concentrations show significant differences between Berkenwoude, De Nesse and the reference area and within the specific areas. After the water level increase, peaks that occurred simultaneously existed within the areas, even in the reference area, where the water level has not been restored. Variation in phosphate levels over time exist for one site in De Nesse and two sites in the reference area. The fact that significant differences before and after June 2017 exist in the reference area indicates that factors other than the water level increase exist that influenced phosphate levels in Krimpenerwaard.

3.2.3 Water transparency

Water transparency in Krimpenerwaard varied between 0.05 m (Berkenwoude, July 2018) and 0.8 m (Reference area, several times between 2004 and 2009; appendix A figure 3). Water is significantly more transparent in the reference area (0.44 ± 0.15 m) than in De Nesse (0.34 ± 0.07 m) and Berkenwoude (0.28 ± 0.09 m, appendix A table 11). In De Nesse, water transparency is significantly higher than in Berkenwoude as well (appendix A table 11).

Berkenwoude

Water transparency at site 1 in Berkenwoude differs significantly from the other three site (appendix A table 12). With a value of 0.33 ± 0.09 m, water was significantly clearer than at site 2 (0.23 ± 0.09 m), site 3 (0.24 ±0.06 m) and site 4 (0.28 ± 0.09 m). The other sites do not differ significantly from each other (appendix A table 12).

Any effects of the water level increase on water transparency are not detected in Berkenwoude (appendix A table 13). Nevertheless, a drop in water transparency is observed at site 2, site 3 and site 4 in July 2018 (figure 3.4).

De Nesse

Water transparency in De Nesse was 0.34 ± 0.08 m, 0.36 ± 0.07 m and 0.33 ± 0.08 m at site 1, site 2 and site 3, respectively. These differences in water transparency are not significant ($X^2 = 4.73$, $p = 0.09$).

Even though differences between sites do not exist, water was significantly more turbid after the water level increase than before at site 2 (before: 0.38 ± 0.07 m, after: 0.32 ± 0.07 m; figure 3.4, appendix A table 13). In contrast, at site 3, water transparency was significantly lower before the water level increase (0.31 ± 0.08 m) than after the water level increase (0.35 ± 0.07 m; appendix A table 13). At site 1, water transparency did not differ significantly before and after the water level increase (appendix A table 13).

Reference area

Water transparency at site 1 of the reference area is significantly higher (0.48 ± 0.16 m) than transparency at site 2 (0.41 ± 0.12 m) and site 3 (0.44 ± 0.15 m) (appendix A table 14). Even though the differences in transparency were thus only a few centimetres, they are still significant. The difference in water transparency between site 2 and site 3 is not significant.

Water transparency did not differ significantly before and after June 2017 at site 1 and site 3 (appendix A table 13). Nonetheless, at site 2, transparency was significantly lower after water levels have been increased in Berkenwoude and De Nesse (0.33 ± 0.07 m) than before this increase (0.42 ± 0.13 m). This implies that factors other than the water level increase may have influenced water transparency in polder Krimpenerwaard.

General patterns in water transparency

Significant differences in water transparency exist between locations and within locations and over time. In Berkenwoude, no effects of the water level increase can be detected. In De Nesse, water transparency differs significantly after the water level increase compared to before the rewetting event at two of the tree sites, but changes are not uniform. At the reference area, variation over time exists as well, indicating that other factors, e.g. temperature, may have played a role in water transparency as well.

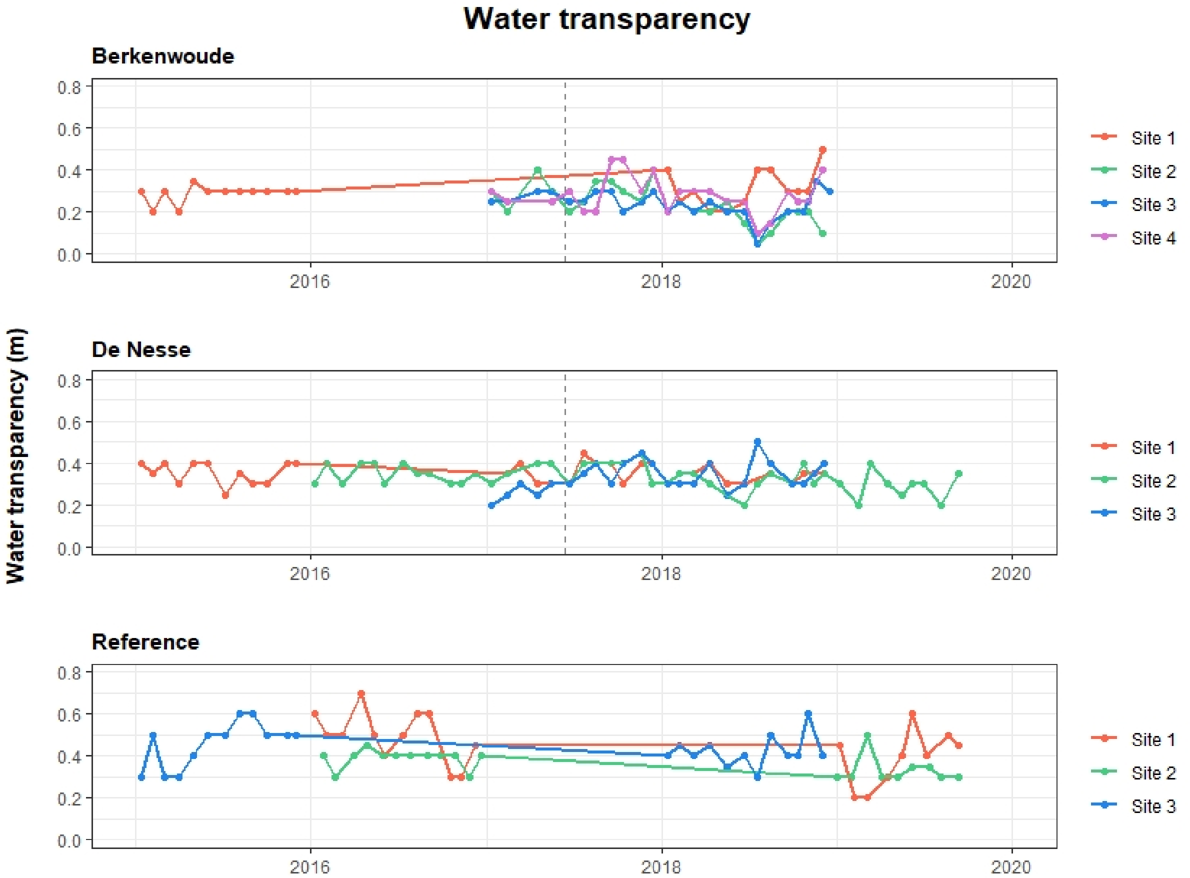

Figure 3.4 Water transparency (m) in Berkenwoude, De Nesse and the reference area. The dashed line indicates the water level increase in Berkenwoude and De Nesse. In the reference area, water levels have not been increased.

3.2.4 General patterns in water quality

In general, the variation in total nitrogen and phosphate concentrations and water transparency is large in polder Krimpenerwaard, both within and between the three different locations.

The increase of the water level in Berkenwoude and De Nesse did not result in a uniform change in water quality in terms of nutrient levels and water transparency. In Berkenwoude, any effects of the water level increase on nutrient levels and water transparency cannot be detected. In De Nesse, significant differences in water quality indicators exist, but changes are not uniform. Phosphate concentrations and water transparency were significantly different after the water level increase than before at two of the tree sites. However, these changes are contradicting: at site 2, both phosphate concentrations and water transparency decreased after the water level increase. At site 3, phosphate concentrations, nitrogen concentrations and water transparency decreased after the water level increase. No differences in phosphate concentrations and water transparency are detected at site 1. However, nitrogen concentrations decreased after the water level increased at site 1.

In the reference area, significant differences before and after June 2017 exist for phosphate concentrations and water transparency. Phosphate levels were significantly lower after June 2017 at site 1 and site 3, water transparency was significantly lower after June 2017 at site 2. These decreases in water quality indicate that other factors such as weather (temperature, precipitation) may have played a role in water quality in Krimpenerwaard as well.

3.3 Terrestrial and aquatic vegetation

Across the three locations – Berkenwoude, De Nesse and the reference area – differences in both terrestrial and aquatic vegetation exist. Moreover, variation in vegetation type and dominant species across monitoring years exist as well.

Berkenwoude

Terrestrial vegetation

Total terrestrial vegetation cover in Berkenwoude was equal at all sample sites and during all years (88%). Moreover, species richness was within the same order of magnitude between sites and over time (appendix B figure 1). However, vegetation type of the six sites on land in Berkenwoude varies (table 3.4). Within the sample sites, vegetation type changed over time as well at most sites (table 3.4). Vegetation type did not change within the monitoring period at site 2 and site 5 (table 3.4). Even though vegetation type did not vary over time at these sites, which indicates that abiotic conditions did not fluctuate, dominant species changed over time at both sites (table 3.4). At both sites, the variation in dominant species is likely to be explained by the rewetting in June 2017. At site 2, water mint (*Mentha aquatica* L.) and *Sparganium erectum s.l.* (simplestem bur-reed or branched bur-reed) were the dominant species in 2018. *S. erectum s.l.* favors wet to temporary tidal soils (Hennekes, Smits, & Schaminee, 2010). Moreover, water mint prefers slightly wetter conditions (Ellenberg class 8) than the species that were dominant before the water level increase (Hennekes et al., 2010). At site 5, the change in dominant species over time direct to a change in soil conditions towards a more nutrient-poor soil that is slightly wetter (Hennekes et al., 2010).

At all other sites, vegetation type (table 3.4) varied over time. At site 1, vegetation types resembled each other regarding nutrient availability, soil acidity and moisture class during the period 2002 – 2015 (Hennekes et al., 2010; appendix B figure 2,3,4). Vegetation type in 2018 is typically characterised by a lower soil fertility and a more acidic soil (Hennekes et al., 2010). The dominant species at site 1 changed from yorkshire fog (*Holcus lanatus* L.) before the water level increase to great manna grass (*Glyceria maxima* (Hartm.) Holmb.) after the water level increase (table 3.4). Great manna grass favors higher moisture levels than yorkshire fog (Hennekes et al., 2010). Therefore, the change in dominant species is likely to be explained by the rewetting event.

The vegetation type that was dominant at site 3 in 2018 is characterised by a somewhat higher soil fertility (ellenberg class 6 compared to ellenberg class 5, appendix B figure 2) than the characteristic vegetation type in 2011 and 2015. Soil acidity and moisture class are similar between the vegetation types (appendix B figure 3 and 4). This higher soil fertility is represented by the dominant species in 2018 as well. Sweet flag (*Acorus calamus* L.), which dominated in 2018, favors a slightly higher soil fertility than water mint, which was dominant during all monitoring years and yorkshire fog, which was dominant in 2011 (table 3.4).

At site 4, the vegetation types and dominant species during the monitoring years indicate that restoration of the water level has not resulted in a landscape that is significantly wetter. Vegetation changed from a vegetation type with moderate to rich soils and moderate acid soils to weakly acid soils to a vegetation type that typically has poor to moderate rich soils that have moderate acidity (Hennekes et al., 2010). Typical water preference of the two vegetation types (table 3.4) is similar (appendix B figure 4). The simultaneous dominance of *Carex oederi s.l.* and *Phragmites australis* (Cav.) Trin. ex Steud. (Common reed) in 2018 is peculiar, as the latter favors nutrient rich soils and weakly acid soils while the former grows on nutrient poor soils with moderate acidity (Hennekes et al., 2010). Soil moisture preference of both species is comparable.

Table 3.4 Overview of the vegetation types (code and name) and the dominant species over time at the different monitoring sites on land in Berkenwoude. Specific cover percentage of the species is given between brackets in the column stating the dominant species.

BERKENWOUDE - TERRESTRIAL				
Monitoring site	**Year**	**Vegetation type (code)**	**Vegetation type (name)**	**Dominant species**
1	2002	16AB04A	Ranunculo-Senecionetum juncetosum articulati	*Holcus lanatus* (38%)
	2007	16RG02	Holcus lanatus-Lychnis flos-cuculi-[Molinietalia]	*Holcus lanatus* (38%)
	2011	12BA02A	Triglochino-Agrostietum cardaminetosum	*Holcus lanatus* (18%)
	2015	12BA02A	Triglochino-Agrostietum cardaminetosum	*Holcus lanatus* (18%)
	2018	16AB04B	Ranunculo-Senecionetum caricetosum paniceae	*Glyceria maxima* (38%)
2	2007	16RG06	Carex disticha-[Calthion palustris]	*Carex disticha* (38%)
	2011	16RG06	Carex disticha-[Calthion palustris]	*Galium palustre (s.l.)* (38%)
	2015	16RG06	Carex disticha-[Calthion palustris]	*Galium palustre (s.l.) – Holcus lanatus* (18%)
	2018	16RG06	Carex disticha-[Calthion palustris]	*Mentha aquatica – Sparganium erectum (s.l.)* (38%)
3	2011	12BA02A	Triglochino-Agrostietum cardaminetosum	*Holcus lanatus – Mentha aquatica* (38%)
	2015	16RG06	Carex disticha-[Calthion palustris]	*Mentha aquatica (63%)*
	2018	29AA01	Polygono-Bidentetum	*Mentha aquatica – Acorus calamus* (38%)
4	2002	12BA02A	Triglochino-Agrostietum cardaminetosum	*Calliergonella cuspidata* (38%)
	2007	16AB04B	Ranunculo-Senecionetum caricetosum paniceae	*Calliergonella cuspidata – Carex oederi* (38%)
	2011	16AB04B	Ranunculo-Senecionetum caricetosum paniceae	*Calliergonella cuspidata* (63%)
	2015	16AB04B	Ranunculo-Senecionetum caricetosum paniceae	*Calliergonella cuspidata* (63%)
	2018	16AB04B	Ranunculo-Senecionetum caricetosum paniceae	*Calliergonella cuspidata - Carex oederi s.l. - phragmites australis* (63%)
5	2011	16AB04B	Ranunculo-Senecionetum caricetosum paniceae	*Anthoxanthum odoratum* (38%)
	2016	16AB04B	Ranunculo-Senecionetum caricetosum paniceae	*Carex oederi s.l.* (38%)
	2018	16AB04B	Ranunculo-Senecionetum caricetosum paniceae	*Calliergonella cuspidata - Carex oederi s.l.* (38%)

Aquatic vegetation

Aquatic species richness was similar between the four different sites in Berkenwoude (appendix B figure 5). Over time, the numer of species seemed to slightly decrease at three of the four monitoring sites (appendix B figure 5). Total vegetation cover of aquatic species varied across the monitoring sites and over time (appendix B figure 6). After the water level increase, total aquatic vegetation cover was lower at all monitoring sites in Berkenwoude than before (appendix B figure 6). At site 1, 7 and 8, aquatic vegetation cover was only 2%, 13% and 13% in 2018, respectively, while in earlier years, vegetation cover was up to 88% (appendix B figure 6). At all four monitoring sites, vegetation type changed over time (table 3.5). Nontheless, preferences regarding fertility and acidity do not show large variation across vegetation types. The dominant species in the ditch and their specific cover show variation as well (table 3.5). However, they are all eutrophic species, duckweeds or algae being dominant at all sites in several monitoring years (table 3.5).

Table 3.5 Overview of the vegetation types (code and name) and the dominant species over time at the different monitoring sites in ditches in Berkenwoude. Specific cover percentage of the species is given between brackets in the column stating the dominant species. NA-strings indicate the absence of a dominant species (specific cover < 8%).

BERKENWOUDE - AQUATIC				
Monitoring site	**Year**	**Vegetation type (code)**	**Vegetation type (name)**	**Dominant species**
1	2002	05BB01	Stratiotetum	*Zygnemataceae* spp. (88%)
	2007	05RG04	Ceratophyllum demersum-[Nupharo-Potametalia]	*Ceratophyllum demersum* (63%)
	2011	05BB01	Stratiotetum	*Lemna gibba – Lemna gibba/minor* (38%)
	2015	05RG04	Ceratophyllum demersum-[Nupharo-Potametalia]	*Ceratophyllum demersum* (38%)
	2018	08BA01	Cicuto-Calletum	NA
6	2002	05RG04	Ceratophyllum demersum-[Nupharo-Potametalia]	*Zygnemataceae* spp. (63%)
	2007	01AA01B	Wolffio-Lemnetum azolletosum filiculoides	*Lemna gibba* (38%)
	2011	05BB01	Stratiotetum	*Zygnemataceae* spp. (38%)
	2015	05BC05	Myriophyllo verticallati-Hottonietum	*Lemna turionifera* (63%)
	2018	05BC05	Myriophyllo verticallati-Hottonietum	NA
7	2007	05RG04	Ceratophyllum demersum-[Nupharo-Potametalia]	*Azolla filiculoides* (88%)
	2011	05RG04	Ceratophyllum demersum-[Nupharo-Potametalia]	*Hydrocharis morsus-ranae* (63%)
	2015	05BA03	Myriophyllo-Nupharetum	*Ceratophyllum demersum – Hydrocharis morsus-ranae – Lemna turionifera* (18%)
	2018	05BC05	Myriophyllo verticallati-Hottonietum	*Nuphar lutea* (8%)
8	2011	05BA03	Myriophyllo-Nupharetum	*Vaucheria* spp. (63%)
	2015	05RG04	Ceratophyllum demersum-[Nupharo-Potametalia]	*Ceratophyllum demersum* (88%)
	2018	05BB01	Stratiotetum	*Nuphar lutea* (38%)

De Nesse

Terrestrial vegetation

At the terrestrial monitoring site in De Nesse, total cover percentage (88%) was stable over time. Total species richness showed a slight increase (appendix B figure 1), but vegetation type and dominant species show little variation over time (table 3.6). Vegetation type was equal in 2003 and after 2011 (table 3.6), which implies stable conditions regarding soil fertility, acidity and wetness. This is partly supported by the dominant species at the site during the monitoring years. The species have similar preferences regarding acidity and fertility, but moisture preferences differ (Hennekes et al., 2010). *Glyceria maxima* and *Carex acuta* (L.) prefer wetter soils than *Holcus lanatus* and *Poa trivialis* (L.)

(appendix B figure 7). The water level increase may explain why *Holcus lanatus* and *Poa trivialis* were not dominant in 2018. On the other hand, *Carex acuta*, which is typical in marshes and wet grasslands, had a lower specific cover (table 3.6 (Hennekes et al., 2010; NDFF, 2007; Peintinger, Prati, & Winkler, 2007).

Table 3.6 Overview of the vegetation types (code and name) and the dominant species over time at the monitoring site on land in De Nesse. Specific cover percentage of the species is given between brackets in the column stating the dominant species.

DE NESSE - TERRESTRIAL				
Monitoring site	**Year**	**Vegetation type (code)**	**Vegetation type (name)**	**Dominant species**
1	2003	12BA02A	Triglochino-Agrostietum cardaminetosum	*Holcus lanatus* (38%)
	2007	16RG02	Holcus lanatus-Lychnis flos-cuculi-[Molinietalia]	*Glyceria maxima – Holcus lanatus – Poa trivialis* (38%)
	2011	12BA02A	Triglochino-Agrostietum cardaminetosum	*Holcus lanatus* (38%)
	2015	12BA02A	Triglochino-Agrostietum cardaminetosum	*Carex acuta – Glyceria maxima – Holcus lanatus* (18%)
	2018	12BA02A	Triglochino-Agrostietum cardaminetosum	*Glyceria maxima* (38%)

Aquatic vegetation

Total aquatic vegetation cover varied between the two sites in De Nesse and over time (appendix B figure 8). Before 2011, vegetation cover was high with a minimum cover of 63%. Hereafter, total aquatic vegetation cover decreased to 13% in 2015 and 2018 at site 1 (appendix B figure 8). At site 3, vegetation cover decreased to 13% in 2015, after which aquatic vegetation cover increased to 88% in 2018 again (appendix B figure 8). Despite the decreases in vegetation cover, species richness remained relatively constant over time (appendix B figure 5). Vegetation types were equal between the two sites in 2018 (table 3.7). Moreover, the vegetation types at the two site highly resembled each other regarding soil fertility and pH (appendix B figure 9 and 10; Hennekes et al., 2010). At site 1 FLAB (Floating Algae Beds) dominated the system at three of the five monitoring years (table 3.7). At site 3, vegetation cover was dominated by algae and duckweed during all monitoring years (table 3.7). This indicates a high nutrient availability in both ditches (table 3.7).

Table 3.7 Overview of the vegetation types (code and name) and the dominant species over time at the different monitoring sites in ditches in De Nesse. Specific cover percentage of the species is given between brackets in the column stating the dominant species. NA-strings indicate the absence of a dominant species (specific cover < 8%).

DE NESSE – AQUATIC				
Monitoring site	**Year**	**Vegetation type (code)**	**Vegetation type (name)**	**Dominant species**
1	2003	05RG04	Ceratophyllum demersum-[Nupharo-Potametalia]	*Zygnemataceae* spp. (63%)
	2007	05RG04	Ceratophyllum demersum-[Nupharo-Potametalia]	*Ceratophyllum demersum* (38%)
	2011	05BB01	Stratiotetum	*Zygnemataceae* spp. (63%)
	2015	05BC05	Myriophyllo verticallati-Hottonietum	*Vaucheria* spp. (8%)

	2018	05BC05	Myriophyllo verticallati-Hottonietum	*Hydrocharis morsus-ranae* (8%)
3	2003	05BB01	Stratiotetum	*Lemna trisulca - Stratiotes aloides - Lemna gibba/minor* (38%)
	2007	05BB01	Stratiotetum	*Lemna gibba* (63%)
	2011	05BC03	Ranunculetum circinati	*Zygnemataceae* spp. (38%)
	2015	05BC05	Myriophyllo verticallati-Hottonietum	*Lemna turionifera* (8%)
	2018	05BB01	Stratiotetum	*Lemna gibba* (38%)

Reference area

Terrestrial vegetation

Whereas total vegetation cover percentage was 88% in both sites for all monitoring years, species richness, vegetation type and dominant species vary within the reference area (table 3.8; appendix B figure 1). However, soil characteristics that are associated with the dominant vegetation types in the reference area only show minor differences (appendix B figure 2, 3 and 4).

Average species richness was with approximately 50 species higher at site 1 than at site 2 (ca. 40 species; appendix B figure 1). Vegetation type at site 1 did not change during the monitoring period (table 3.8). Nonetheless, the dominant species changed over time, even though species specific soil preferences are similar for all species, except for *Sparganium erectum* s.l, who prefers wet to temporary tidal soils (Hennekes et al., 2010; appendix B figure 7). The other dominant species prefer dry/moist to moist soils (Hennekes et al., 2010; appendix B figure 7).

The three different vegetation types that prevailed at site 2 during the monitoring period only slightly differ from each other concerning soil fertility, acidity and moisture (table 3.8; appendix B figure 2, 3 and 4). The little variation in dominant species over time supports this (table 3.8). However, soil moisture preferences of Yorkshire fog (dominant between 2007 and 2015) and red fescue (dominant in 2018) are slightly different, red fescue preferring slightly dryer soil conditions (Hennekes et al., 2010). This indicates that in the reference area, where water levels have not been increased in 2017, the soil dried.

Table 3.8 Overview of the vegetation types (code and name) and the dominant species over time at the different monitoring sites on land in the reference area. Specific cover percentage of the species is given between brackets in the column stating the dominant species.

REFERENCE AREA – TERRESTRIAL				
Monitoring site	**Year**	**Vegetation type (code)**	**Vegetation type (name)**	**Dominant species**
1	2007	12BA02A	Triglochino-Agrostietum cardaminetosum	*Calliergonella cuspidata* (38%)
	2011	12BA02A	Triglochino-Agrostietum cardaminetosum	*Holcus lanatus / Ranunculus repens / Sparganium erectum s.l. / Brachythecium rutabulum / Calliergonella cuspidata* (18%)
	2015	12BA02A	Triglochino-Agrostietum cardaminetosum	*Holcus lanatus* (18%)
	2018	12BA02A	Triglochino-Agrostietum cardaminetosum	*Agrostis stolonifera* (63%)

2	2007	16RG02	Holcus lanatus-Lychnis flos-cuculi-[Molinietalia]	*Holcus lanatus* (63%)
	2011	16AB04A	Ranunculo-Senecionetum juncetosum articulati	*Holcus lanatus* (63%)
	2015	16RG02	Holcus lanatus-Lychnis flos-cuculi-[Molinietalia]	*Holcus lanatus* (63%)
	2018	32AA01B	Valeriano-Filipenduletum holcetosum	*Festuca rubra* (63%)

Aquatic vegetation

Total vegetation cover at the monitoring sites in the reference area was high during the entire length of the monitoring period, except for site 4 in 2018. Here, vegetation cover was 13%, while vegetation cover was up to 88% at the other three sites (appendix B figure 11). At site 2 and 3, total vegetation cover was 88% during all monitoring years. At site 1, total cover was only lower (38%) in 2018. Whereas at the monitoring sites in Berkenwoude and De Nesse, vegetation cover was dominated by duckweeds and algae only, in the reference area hornwort (*Ceratophyllum demersum* L.) reached high cover percentages up to 88% as well, indicating a water transparency that is sufficient for macrophytes to grow (figure 3.9; Ejankowski & Solis, 2014). However, vegetation types are equal to the vegetation types observed in Berkenwoude and De Nesse, indicating no differences in soil fertility and acidity (figure 3.5, 3.7 and 3.9). Species richness shows some variation between the four monitoring sites and over time (appendix B figure 5).

Table 3.9 Overview of the vegetation types (code and name) and the dominant species over time at the different monitoring sites in ditches in the reference area. Specific cover percentage of the species is given between brackets in the column stating the dominant species. NA-strings indicate the absence of a dominant species (specific cover < 8%).

REFERENCE AREA - AQUATIC				
Monitoring site	**Year**	**Vegetation type (code)**	**Vegetation type (name)**	**Dominant species**
1	2007	05RG04	Ceratophyllum demersum-[Nupharo-Potametalia]	*Zygnemataceae* spp. (63%)
	2011	05RG04	Ceratophyllum demersum-[Nupharo-Potametalia]	*Ceratophyllum demersum – Lemna gibba/minor* (18%)
	2015	05RG04	Ceratophyllum demersum-[Nupharo-Potametalia]	*Ceratophyllum demersum* (63%)
	2018	05BB01	Stratiotetum	*Ceratophyllum demersum*
2	2007	05RG04	Ceratophyllum demersum-[Nupharo-Potametalia]	*Ceratophyllum demersum – Lemna gibba/minor* (63%)
	2011	05RG04	Ceratophyllum demersum-[Nupharo-Potametalia]	*Ceratophyllum demersum – Zygnemataceae* spp. (38%)
	2015	05RG04	Ceratophyllum demersum-[Nupharo-Potametalia]	*Ceratophyllum demersum – Zygnemataceae* spp. (88%)
	2018	05BC05	Myriophyllo verticallati-Hottonietum	*Hydrocharis morsus-ranae* (18%)
3	2007	05BB02	Utricularietum vulgaris	*Spirodela polyrhiza – Utricularia vulgaris* (38%)
	2011	05RG04	Ceratophyllum demersum-[Nupharo-Potametalia]	*Lemna trisulca* (38%)
	2015	01AA02A	Lemno-Spirodeletum typicum	*Lemna trisulca* (38%)

	2018	05BB01	Stratiotetum	*Spirodela polyrhiza* (38%) (88%)
4	2007	05RG04	Ceratophyllum demersum-[Nupharo-Potametalia]	*Ceratophyllum demersum* (88%)
	2011	05RG04	Ceratophyllum demersum-[Nupharo-Potametalia]	*Ceratophyllum demersum* (63%)
	2015	05BB02	Utricularietum vulgaris	*Utricularia vulgaris – Vaucheria* spp. (18%)
	2018	05BC05	Myriophyllo verticallati-Hottonietum	*Hydrocharis morsus-ranae – Lemna gibba* (8%)

Differences in vegetation composition between polders

Terrestrial vegetation

Terrestrial vegetation throughout Berkenwoude, De Nesse and the reference area do not show large differences. Total vegetation cover was high at all sites during all years and species richness is in the same order of magnitude. Of the fourteen different vegetation types that were present at the different sites during the monitoring period, only two were present in all three polders and only one was present in two of the three polders. Nonetheless, vegetation types did not show large differences regarding soil fertility and acidity (appendix B figure 2 and 3). Regarding moisture levels, vegetation types that were dominant in Berkenwoude and De Nesse after the water level increase seem slightly wetter than vegetation types that were dominant before the water level increase (appendix B figure 4).

The dominant species at the three locations show large variation in moisture preference (appendix B figure 7). Moisture preferences of the species vary between dry/moist soils to moist soils (Ellenberg moisture class 6) to temporary tidal water (Ellenberg moisture class 10) (Hennekes et al., 2010). Species that were dominant in 2018 in Berkenwoude and De Nesse are typically species that prefer high moisture levels (appendix B figure 7).

Aquatic vegetation

Aquatic vegetation composition and species richness is comparable between Berkenwoude, De Nesse and the reference area. Only nine different vegetation types were present, of which three were present in all areas during the monitoring period. The vegetation types that were present in Berkenwoude and De Nesse after the water level increase are characterised by a slightly lower soil fertility (appendix B figure 9).

Duckweed and algae composed the main vegetation in all three areas and in several monitoring years (figure 3.5, 3.7 and 3.9). All three locations are characterised by a decrease in total vegetation cover during the monitoring period. Nevertheless, in De Nesse and the reference area, vegetation coverage remained high (88%) at some sites (appendix B figure 8 and 11). In Berkenwoude, total aquatic vegetation cover was lower than or equal to 38% at all four monitoring plots.

4. DISCUSSION

4.1 Meadow bird territories

4.1.1 General effects of a water level increase on meadow bird diversity and abundance

An increase in heterogeneity of the landscape and an increase in food availability are the two main mechanisms explaining the positive effects of water level increases on meadow birds (Benton, Vickery, & Wilson, 2003; Green, Hirons, & Cresswell, 1990; Verhulst et al., 2007). Higher water levels can lead to more variation in vegetation structure and -diversity throughout the landscape, primarily due to local elevation differences and small-scale periodic flooding (Görn, Schulze, & Fischer, 2015; Zmihorski et al., 2016). As a result of this increase in heterogeneity, more shelter and nesting sites for meadow birds are created (Davis, 2005; Vandenberghe, Prior, Littlewood, Brooker, & Pakeman, 2009; Warren & Anderson, 2005). In addition, the increased vegetation structure and -diversity can have an effect on the abundance and composition of insect communities (Dennis, Young, & Gordon, 1998; Olivier, Aranda, Godoi, & Gracioll, 2014; Romero-Alcaraz & Ávila, 2000; Visser, Melman, Buij, & Schotman, 2017). As meadow passerines and wader chicks are relying on insects as a food source, water level increases can positively affect these bird groups.

The waders that are characteristic for meadows rely on soil biota (e.g. annelids, larvae of crane flies (leatherjackets)) as a food source (Green et al., 1990; Visser et al., 2017). Wetness of soils is vital for wader bird foraging in two ways (Green et al., 1990; Zmihorski et al., 2016). First, soil biota require wet or moist conditions, depending on the species (Rhymer, Robinson, Smart, & Whittingham, 2010; Riggins, Davis, & Hoback, 2009). Soil dryness either forces soil invertebrates deeper into the bottom or is directly lethal to them, leading to food scarcity for wading birds (Rhymer et al., 2010). Wetness thus increases the species abundance of soil biota. As the rewetting event results in heterogeneity in soil wetness, the abundance of invertebrate species ranging from a variety of soil moisture preferences is likely to increase (Davis, Austin, & Buhl, 2006). Second, wetness of soils improves the penetrability of the soil, thereby increasing the accessibility of soil invertebrates for tactile feeding (Green et al., 1990; Zmihorski et al., 2016).

Indirectly, water level increases can affect nest or chick predation. As a consequence of increased wetness, predation of meadow birds by land predators is impeded, leading to lower predation rates (Teunissen, Schekkerman, & Willems, 2005). Moreover, increases in vegetation structure that may be the result of increased water levels provide more protection than homogenous grasslands, thereby decreasing meadow bird predation by predatory birds or rats (Schekkerman, Teunissen, & Oosterveld, 2009).

4.1.2 Bird patterns observed in Krimpenerwaard

Meadow bird species compositions were similar across Berkenwoude, De Nesse and reference polder Bilwijk. The LPI decreased in Bilwijk, while after the water level increase in June 2017, the LPI showed an increase in Berkenwoude and De Nesse. The increase in LPI in Berkenwoude and De Nesse can be explained by the two (coexisting), above stated mechanisms: the landscape may have become more favourable for meadow birds and food availability may have increased as a result of the water level increase (e.g. Davis, 2005; Görn et al., 2015; Green et al., 1990; Zmihorski et al., 2016).

In 2018, meadow bird abundance per 100 ha was almost twice as high in De Nesse than in Berkenwoude, even though the LPI of Berkenwoude and De Nesse both increased after the water level increase. This substantial abundance difference between the rewetted polders De Nesse and Berkenwoude may be explained by the structure of the polders. De Nesse is a much more open polder than Berkenwoude, which is characterised by several small forest patches. Meadow bird species prefer open grasslands, which may explain the much higher densities of meadow birds in De Nesse compared to Berkenwoude (Burger, Burger, & Faaborg, 1994; Helzer, 1996; Helzer & Jelinski, 1999). Moreover, taking into account the edge effect – physical and biotic alterations in the zone next to the edge of a habitat (Laurance et al., 2007) – the area of suitable meadow bird habitat in Berkenwoude is effectively much smaller than the area size of polder Berkenwoude.

The higher meadow bird densities after rewetting are in accordance with other rewetted areas (e.g. Belting, 2007; Blüml, 2012; Gerritsen & Roodhart, 2018; Görn et al., 2015; Mischenko, Sukhanova, & Zöckler, 2014; Zmihorski et al., 2016). A literature study did not result in studies claiming a decrease in abundances after rewetting. The largest experiment regarding the effects of a water level increase on meadow bird diversity has been executed in Dümmer (Germany) (Belting, 2007). Here, a large area of 2500 ha has been rewetted between 1998 and 2007. As a result, a suitable habitat was created for numerous meadow bird species, which resulted in a strong increase in bird populations that still continues (Belting, 2007). In addition, eleven species that had disappeared from the area since dikes were built in 1953, returned after the area was rewetted (Belting, 2007). This implies that restoration of the water levels in the entire polder Krimpenerwaard has a great potential for meadow bird conservation.

Bird monitoring in Berkenwoude and De Nesse was executed in spring 2019 again (Van Donk et al., 2019). This study revealed that bird diversity increased in 2019 compared to 2017 and 2018, indicating that bird diversity after rewetting is still increasing (Van der Winden et al., 2018; Van der Winden, Kanters, Poot, & Van Horssen, 2017; Van Donk et al., 2019). The short-term trend in increasing meadow bird diversity after rewetting is thus in line with the expected long-term biodiversity trends after rewetting.

We must bear in mind that only meadow bird monitoring data of one year after the water level increase existed for this study. As a result, Berkenwoude and De Nesse are still in the process of changing, which may affect meadow bird diversity. Moreover, as indicated by the rewetting in Dümmer, it may take several years for species to (re-)establish in the rewetted polder (Belting, 2007). Continued monitoring is required to gain information about the long-term changes in the polder that are associated with the water level increase.

4.2 Water quality

4.2.1 General effects of a water level increase on water quality

Agricultural areas that are transformed to nature reserves are generally eutrophic as a result of former, prolonged fertiliser use (Berendse, Oomes, Altena, & Elberse, 1992; Kieckbusch & Schrautzer, 2007). Soil nitrates are released quickly by evaporation through the process of denitrification or by the run off to groundwater and surface waters (Lamers, Lucassen, Smolders, & Roelofs, 2005). As a consequence, they rapidly disappear from the system (Kieckbusch & Schrautzer, 2007; Lamers et al., 2005).

In contrast, phosphate that is inherited from the agricultural history remains bound to the soil when soils are drained (Kieckbusch & Schrautzer, 2007; Lamers et al., 2005; Olde Venterink, Davidsson, Kiehl, & Leonardson, 2002). When water levels are raised, phosphate that is bound to iron is mobilised, as iron is reduced from Fe^{3+} to Fe^{2+}, which has a lower binding capacity (Lamers et al., 2005; Zak, Gelbrecht, & Steinberg, 2004). This process of phosphate mobilisation can be amplified if surface- and groundwaters are rich in sulphate (Lamers et al., 2005; Roelofs, 1991; Zak & Gelbrecht, 2007). As a consequence of the water level increase, this sulphate can be reduced to sulphide, which leads to additional phosphate being mobilised (Lamers et al., 2005; Roelofs, 1991; Zak & Gelbrecht, 2007). The mobilised phosphate causes internal eutrophication, which may cause a system to shift to its alternative (turbid) stable state (Lamers et al., 2005; Olde Venterink et al., 2002; Scheffer, Hosper, Meijer, Moss, & Jeppesen, 1993). As the process of phosphate mobilisation can last decades to centuries (Lamers et al., 2005), nutrient levels can remain high for a long time.

Nevertheless, if soil iron concentrations are high, the opposite is observed (Lamers et al., 2013; Vonk et al., 2017; Zak et al., 2004). If Fe/P ratios are larger than approximately 3 (Zak et al., 2004), phosphate that is released from the soil binds to iron and precipitates, resulting in a decrease in phosphate levels (Vonk et al., 2017; Zak et al., 2004). The dominant process, either release of phosphate or precipitation of phosphate, determines whether ditches eutrophicate after rewetting or not.

4.2.2 Patterns observed in Krimpenerwaard

In Krimpenerwaard, large variation in phosphate, total nitrogen and water transparency existed between and within the three investigated locations. Moreover, the patterns found in the three variables are contradicting. In the reference area, both phosphate concentrations and water transparency were significantly higher than in De Nesse and Berkenwoude. This is striking as high nutrient levels are generally associated with low water transparency (Scheffer et al., 1993).

The fact that phosphate concentrations have been found to be higher in the reference area than in Berkenwoude and De Nesse indicates that the expected increase in phosphate levels due to phosphate mobilisation after rewetting remained absent. Moreover, any differences in water transparency and nutrient concentrations before and after the water level increase did not exist in Berkenwoude. This contradicts the hypothesis that the water level increase led to an increase in phosphate levels. In addition, a decrease in phosphate was neither present, indicating that precipitation of phosphate did neither play a role.

In De Nesse, both phosphate and transparency decreased after the water level increase and nitrogen remained stable. The decrease in phosphate levels may be the result of phosphate precipitation due to a binding with iron. Research by Van Gerven et al. (2011) indicates that iron levels in Krimpenerwaard are likely to be sufficient for phosphate to precipitate. However, as their research was performed almost nine years ago, iron levels should be measured again to investigate whether this is still true or whether other processes play a role. Moreover, dredging of the ditches may have played a role in the decrease in phosphate levels as well. The decrease in water transparency implies that factors other than nutrient levels have affected transparency. These factors may include dredging, weather influences and increased sediment resuspension (Valipour, Boegman, Bouffard, & Rao, 2017).

Even though no uniform effects of the water level increase were found in Krimpenerwaard, peaks in phosphate were present after the water level increase, mainly in late spring. This is comparable with a study performed in Germany (Kieckbusch & Schrautzer, 2007). Here, peaks in phosphate existed during summer (Kieckbusch & Schrautzer, 2007). In addition, the absence of an increase in phosphate levels after the water level increase was observed in another study by Vonk et al. (2017) as well.

Overall, the water level increase did not automatically result in an increase in aquatic nutrient concentrations and a decrease in water transparency. Nevertheless, aquatic vegetation cover was found to decrease (see section 3.3), even though water transparency had not changed in Berkenwoude after the water level increase. This indicates that growth of macrophytes was affected by other factors. Moreover, water transparency was not directly related to phosphate levels, as in De Nesse, the water became more turbid, while phosphate levels decreased. This implies that other factors determined turbidity as well. As monitoring was performed once per month during the entire year, effects of seasonality on nutrient levels and turbidity cannot have caused the differences in turbidity and nutrient concentrations.

One must bear in mind that the relatively small sample size may have led to a skewed overview of the phosphate and nitrogen levels and water transparency, as these variables showed large variation in space and time. At most sites, the monitoring period after the water level increase was shorter than the monitoring period before the water level increase, which may have affected the results on the differences between nutrient levels and water transparency before and after the water level increase. Moreover, the time since the water level increase may have been too short for effects to become apparent. To overcome these limitations, continuing monitoring at existing sites and expanding the monitoring at new sites is recommended.

4.3 Terrestrial and aquatic vegetation

4.3.1 Mechanisms underlying the general effects of a water level increase on vegetation composition

Terrestrial vegetation

The prevailing abiotic conditions at a site shape the vegetation composition. Moisture level is one of the primary determinants of vegetation composition (Castelli, Chambers, & Tausch, 2000), with frequency and abundancy of several plant species being directly linked to water levels (Henszey, Pfeiffer, & Keough, 2004).

Preferred moisture levels are highly variable across species, some species being able to survive under a wide range of moisture levels, while other species are only present within narrow boundaries (Dwire, Kauffman, & Baham, 2006). As a consequence, in grasslands having similar soil fertility and -acidity, species compositions and vegetation types may resemble each other under varying moisture levels (Dwire et al., 2006). Nevertheless, the opposite is true as well: multiple vegetation types can be present under similar moisture levels (Hammersmark, Rains, Wickland, & Mount, 2009; Hennekes et al., 2010). This indicates that variables other than moisture levels, such as grazing or mowing and competition between species, determine vegetation composition as well (Hammersmark et al., 2009; Henszey et al., 2004; Perry, Galatowitsch, & Rosen, 2004). In addition, after restoration of the water level, some time is required for the vegetation to adjust to the new hydrologic conditions (Hammersmark et al., 2009). This results in a delay in the manifestation of the effects of an increase in water levels in the vegetation. Effects generally become apparent within a three to six years after rewetting (Hammersmark et al., 2009; Oomes, Olff, & Altena, 1996; Toogood & Joyce, 2009). This time frame allows annual, biennial and many perennial herb species to complete the reproductive cycles a few times and to adjust to the new hydrologic regime (Hammersmark et al., 2009).

It is generally accepted that an increase in water levels can lead to mobilisation of phosphorus that was formerly bound to soil particles (Dieter, Herzog, & Hupfer, 2015; Kieckbusch & Schrautzer, 2007). As a consequence, water level restoration can result in internal eutrophication. In contrast, in iron-rich soils and under aerobic conditions, phosphorus is able to bind to iron and consequently precipitates, resulting in a decrease in phosphate concentrations (Zak et al., 2004). Because of either the mobilisation or the precipitation of phosphate, soil nutrient conditions may change in addition to the water levels, thereby affecting vegetation types as well.

Aquatic vegetation

Increasing water levels may lead to reduction of sulphate, which results in the accumulation of sulphide (Koch, Mendelssohn, & McKee, 1990; Lamers et al., 2013). Sulphide can hamper nutrient uptake, hence limiting growth and development of macrophytes (Koch et al., 1990; Lamers et al., 2013). Moreover, as mentioned above, increasing the water level affects phosphorus availability by either mobilizing it or by precipitating it (Dieter et al., 2015; Kieckbusch & Schrautzer, 2007; Zak et al., 2004). Sediment exposure history – whether the sediment has been permanently inundated or periodically desiccated – is known to affect nutrient fluxes in response to a water level increase as well (Steinman, Ogdahl, Weinert, & Uzarski, 2014; Vonk et al., 2017). Areas that have a history of temporary exposure to air have higher soil nutrient release than soils that were continuously inundated (Steinman et al., 2014; Vonk et al., 2017). As a consequence of its effect on nutrient levels, restoring the water level can either result in more favourable conditions for macrophyte growth, or lead to undesired conditions with algal blooms or dense duckweed cover (Dieter et al., 2015; Vonk et al., 2017).

4.3.2 Vegetation patterns observed in Krimpenerwaard

Terrestrial vegetation

Vegetation types that were dominant in Berkenwoude and De Nesse after the rewetting are associated with slightly higher soil wetness than the vegetation types that were observed in the polders before rewetting. Moreover, species that were dominant after the water level increase favour higher levels of wetness than species that were dominant before the water level increase. This indicates that even though only one year of monitoring data exists after the rewetting, alterations in vegetation composition already become apparent. As the effects of an increase in water level are generally apparent between three to six years (Hammersmark et al., 2009; Oomes et al., 1996; Toogood & Joyce, 2009), and as 2018 was within the 5% driest years since 1906 (KNMI, 2018), the effects of the water level increase on the vegetation composition that are currently observed in Krimpenerwaard are likely to become more apparent in the coming years. The monitoring data indicate that in De Nesse, *Glyceria maxima, Carex* spp. and *Juncus effusus* are likely to become more apparent in the next few years. The monitoring data for Berkenwoude indicates that *Glyceria maxima* and *Juncus effusus* are likely to decrease, while the abundance of *Carex* spp. seem to increase or stabilise.

In addition to the direct relation between higher water levels and vegetation composition, the increase in water levels alters vegetation composition as well. As a consequence of small-scale periodic flooding, which is the result of the higher water levels, the start of the growing season has been postponed, influencing vegetation development (Klimešová, 1994). Moreover, mowing regime had to change as well as mowing is not possible if meadows are flooded. More frequent mowing that starts later in the season is aimed (Kleinjan, pers. comm). As a consequence, species composition is likely to change towards a community that is (relatively) tolerant to mowing (Toogood & Joyce, 2009).

In several other meadow areas, both in the Netherlands (Hoek & Kemmers, 1998; Oomes et al., 1996; Van Dijk, 2008) and in other countries (Henszey et al., 2004; Mauchamp, Chauvelon, & Grillas, 2002; Timmermann et al., 2006; Toogood & Joyce, 2009), experiments with raising water levels have been performed. In all experiments, restoration of the water level resulted in a change in vegetation towards a composition that was indicative for wet conditions (e.g. Oomes et al., 1996; Timmermann et al., 2006; Toogood & Joyce, 2009). Oomes et al. (1996) found a main increase in *Juncus effusus*, while *Carex* spp. highly increased in the rewetted areas in the research of Timmermann et al. (2006).

One year after the water level increase, nutrient richness was comparable to the nutrient levels before the water level increase in Berkenwoude and De Nesse. However, some studies did find an effect of a water level increase on phosphate levels: Van Dijk (2008) found that restoration of the water level led to severe eutrophication, while Oomes et al. (1996) and Hoek & Kemmers (1998) found a decrease in phosphate levels. This differences may be explained by the soil type: the severe eutrophication was found on a peat soil (Van Dijk, 2008), while the decrease in phosphate levels occurred on peaty clay soils (Hoek & Kemmers, 1998; Oomes et al., 1996).

It should be noted that several limitations exist regarding the vegetation composition analyses. First, results of this research are based on only six monitoring sites within the polders Berkenwoude (5) and De Nesse (1). Size of the monitoring sites varied considerably, which may have affected species richness in the plots as well, as plots with a larger size are likely to contain more species. Moreover, aquatic nutrient levels are high in the entire polder Krimpenerwaard (Kleinjan, pers. comm.), which may have affected vegetation composition as well. Last, data on the vegetation composition after the water level increase only exists for one year after the water level increase. As it takes up to six years for effects to become apparent, the current period is too short to draw strong conclusions on the effects of a water level increase on the vegetation composition.

Aquatic vegetation

Vegetation composition was generally similar in Berkenwoude, De Nesse and the reference area. After the water level increase, vegetation types that were present in Berkenwoude and De Nesse are characterised by a slightly lower soil fertility than the vegetation types that were present before the water level increase. This indicates that soil fertility slightly decreased, which may be the result of precipitation of phosphate after rewetting due to a reaction with iron (Lamers et al., 2013; Vonk et al., 2017). Nevertheless, total cover percentage decreased over the years. This decrease in total cover percentage may be caused by several factors. First, it may be the result of the accumulation of sulphide, which impedes nutrient uptake (Lamers et al., 2013; Vonk et al., 2017). Herbivory by waterfowl or invasive red swamp crayfish forms another possible cause for the decrease in vegetation cover (Feminella & Resh, 1989; Fredrickson & Reid, 1988; Lemmers, Crombaghs, & Leuven, 2018; Rodríguez, Bécares, Fernández-Aláez, & Fernández-Aláez, 2005). Due to the burrowing behaviour of red swamp crayfish, they can increase nutrient availability in the water column as well (Angeler, Sánchez-Carrillo, García, & Alvarez-Cobelas, 2001; van der Wal et al., 2013). Last, an increase in flow rates in Berkenwoude and De Nesse may explain the decrease in duckweed cover as duckweed cannot survive in ditches with flow rates exceeding 0.3 metres per second (Leng, Stambolie, & Bell, 1995).

Comparable to this study, restoring the water levels in Oostvaardersplassen (Flevoland, the Netherlands), did not result in an increase in phosphate (Vonk et al., 2017). Nonetheless, restoring water levels can result in nutrient mobilisation in grasslands, which can cause eutrophication if these nutrients runoff to ditches (Baldwin & Mitchell, 2000; Lamers, Lucassen, Smolders, & Roelofs, 2005; Smolders et al., 2019). Possibly, soil type plays a key role in the effects of a water level increase on phosphate levels (Hoek & Kemmers, 1998; Van Dijk, 2008; Vonk et al., 2017).

Limitations regarding the aquatic vegetation are similar to the limitations concerning terrestrial vegetation: 1) the results of this study are based on a limited number of monitoring sites that varied in size in Berkenwoude, De Nesse and the reference area, 2) water transparency is low and nutrient levels are high in Krimpenerwaard, which may have affected macrophyte growth (Kleinjan, pers. comm.; Schep & Verbeek, 2018) and 3) effects of a water level increase may take up to six years to become apparent, indicating that in coming years patterns may emerge that did not exist yet in 2018.

5. CONCLUSIONS

Several conclusions regarding the correlations between the water level increase in Berkenwoude and De Nesse (Krimpenerwaard) and meadow bird diversity, water quality and vegetation composition can be drawn from this research.

The first sub-question was: "*How does an increase in water level affects the biodiversity of meadow birds in terms of species richness and abundance?".* Meadow bird diversity showed an increase in the two rewetted polders, while diversity decreased in the reference area. A literature study revealed that this increase in meadow bird diversity may be the result of two (coexisting) mechanisms. First, the landscape may have become more favourable for meadow birds as a result of a greater variation in vegetation structure and -diversity throughout the landscape after the water level increase. Second, food availability may have increased after rewetting. Therefore, the hypotheses that rewetting results in a meadow bird increase and that food availability is an important mechanism explaining the increase were supported.

The meadow bird abundance per 100 ha after the water level increase was almost twice as high in De Nesse than in Berkenwoude. Literature revealed that the difference may be explained by the structure of the polders. Berkenwoude is a more closed polder that has several small forest patches. As meadow birds prefer a more open structure, this could explain the higher meadow bird abundance in De Nesse than in Berkenwoude. Moreover, due to the small forest patches, the edge effect played a role in Berkenwoude. Consequently, the area of suitable meadow bird habitat in Berkenwoude was effectively much smaller than the total area size of the polder.

To gain insight in the relation between a water level increase and water quality, the following sub-question was formulated: "*To what extent does an increase in water level affect ditch water quality in terms of nutrient concentrations and water transparency?*". The literature study revealed that soil nitrates rapidly evaporate through denitrification or run-off to the groundwater and surface water. This may cause a short increase in ditch nitrogen levels, after which former levels are restored. However, a water level increase generally leads to the mobilisation of phosphate which causes eutrophication. This can cause a decrease in water transparency. However, if the ratio between iron and phosphate is larger than approximately three, phosphate precipitates, leading to a decrease in phosphate levels.

In Krimpenerwaard, large variation in phosphate, total nitrogen and water transparency existed between and within Berkenwoude, De Nesse and the reference area. Phosphate levels and water transparency were significantly higher in the reference area than in Berkenwoude and De Nesse, indicating that an increase in phosphate levels due to phosphate mobilisation after rewetting remained absent. This was supported by the fact that any differences in water transparency and nutrient concentrations before and after the water level increase did not exist in Berkenwoude. In De Nesse, both phosphate concentrations and transparency decreased after the water level increase. Total nitrogen levels remained stable. Therefore, the hypothesis that the increase in water levels led to a significant increase in nutrient concentrations was rejected. The decrease in phosphate levels indicated that phosphate may have been bound to iron in De Nesse. As water transparency in De Nesse decreased after the rewetting while phosphate levels decreased, other factors such as sediment resuspension may have played a role as well. The hypothesis that the water level increase resulted in a decrease in water transparency due to increased nutrient availability was rejected.

To investigate the relation between a water level increase and vegetation composition, the third sub-question was: "*How does an increase in water level affect vegetation composition?".* The literature research showed that rewetting does not only lead to a terrestrial vegetation change towards species that prefer a higher moisture content, it can result in a more diverse vegetation composition as well. Nevertheless, vegetation was found to likely change to species that are more resistant against highly eutrophic conditions as well, as phosphate levels were found to generally increase after rewetting. Regarding aquatic vegetation, the literature study revealed that rewetting results in a decrease in vegetation cover due to the accumulation toxic sulphides and increased eutrophic conditions.

The rewetting event led to a slight difference in terrestrial vegetation type and dominant species in Berkenwoude and De Nesse towards types and species that are associated with slightly higher soil wetness. Species richness remained similar. Therefore, the hypotheses that no differences in vegetation composition existed before and after the water level increase and that richness increased due to the establishment of some water-preferring species after the water level increase were rejected. As monitoring data was available for only one year after rewetting and as the literature study showed that the effects of rewetting on vegetation generally become apparent within six years, the effects of rewetting are likely to become more obvious in the coming years.

Total cover percentage of aquatic vegetation decreased over time in Berkenwoude and De Nesse. Species richness remained similar. Vegetation types that were present in Berkenwoude and De Nesse after the water level increase are characterised by a slightly lower soil fertility than before the increase.

6. RECOMMENDATIONS

For future research, three main recommendations are proposed. First, the sample size should be increased, both in terms of the number of sampling sites and in terms of time. By increasing the sample size, more reliable conclusions can be drawn as any local effects do not have a disproportionate effect on the analyses. Secondly, I recommend investigating causal relations between the water level increase and vegetation composition and water quality in terms of nutrient concentrations and water transparency. This helps to gain more insight in the effectivity of a water level increase as a measure to halt the biodiversity decline in the Netherlands and it helps to create support for the implementation of this measure in other areas. Last, the long-term development of vegetation in all permanent quadrate (PQ) plots in Krimpenerwaard should be investigated to get a better understanding of the observed patterns and to be able to predict outcomes of a water level increase in other areas within and outside polder Krimpenerwaard.

7. REFERENCES

Angeler, D. G., Sánchez-Carrillo, S., García, G., & Alvarez-Cobelas, M. (2001). The influence of Procambarus clarkii (Cambaridae, Decapoda) on water quality and sediment characteristics in a Spanish floodplain wetland. *Hydrobiologia*, *464*, 89–98.

Baldwin, D. S., & Mitchell, A. M. (2000). The effects of drying and re-flooding on the sediment and soil nutrient dynamics of lowland river–floodplain systems: a synthesis. *Regulated Rivers: Research & Management*, *16*(5), 457–467.

Belting, H. (2007). *for the Rewetting of Lake Dümmer Lowlands Niedersachsen*.

Benton, T. G., Vickery, J. A., & Wilson, J. D. (2003). Farmland biodiversity: Is habitat heterogeneity the key? *Trends in Ecology and Evolution*, *18*(4), 182–188.

Berendse, F., Oomes, M. J. M., Altena, H. J., & Elberse, W. T. (1992). Experiments on the restoration of species-rich meadows in The Netherlands. *Biological Conservation*, *62*(1), 59–65.

Besnard, A. A. G., Jeunesse, I. La, Pays, O., Secondi, J., Diversity, S., July, N., & Wiley, P. (2019). Topographic wetness index predicts the occurrence of bird species in floodplains. *Diversity and Distributions 19*(7), 955–963.

BIJ12. (n.d.). N13.01 Vochtig weidevogelgrasland. Retrieved from https://www.bij12.nl/onderwerpen/natuur-en-landschap/index-natuur-en-landschap/natuurtypen/n13-vogelgraslanden/n13-01-vochtig-weidevogelgrasland/ on 03-11-2019

Bijlmer, A. M., & Stempher, W. (2018). *INRICHTINGSPLAN KRIMPENERWAARD*.

Blomqvist, M. M., Vos, P., Klinkhamer, P. G. L., & Ter Keurs, W. J. (2003). Declining plant species richness of grassland ditch banks - a problem of colonisation or extinction. *Biological Conservation*, *109*, 391–406.

Blüml, V. (2012). *Erfolgreiche Feuchtgrünlandentwicklung durch Naturschutzmaßnahmen: langfristige Veränderungen von Flora, Vegetation und Avifauna am Beispiel des Ochsenmoores in der Dümmerniederung*. *Informationdienst Naturschutz Niedersachsen* (Vol. 32). Hannover: NLWKN, Niedersächsischer Landesbetrieb für Wasserwirtschaft, Küsten- und Naturschutz.

Bobbink, R., Hart, M., Van kempen, M., Smolders, F., & Roelofs, J. (2007). *Grondwaterkwaliteitsaspecten bij vernatting van verdroogde natte natuurparels in Noord-Brabant*. Nijmegen.

Bobbink, R., Hornung, M., & Roelofs, J. G. M. (1998). The effects of air-borne nitrogen pollutants on species diversity in natural and semi-natural European vegetation. *Journal of Ecology*, *86*, 717–738.

Bobbink, Roland, & Willems, J. H. (1993). Restoration management of abandoned chalk grassland in the Netherlands. *Biodiversity and Conservation*, *2*(6), 616–626.

Bradbury, R. B., & Kirby, W. B. (2006). Farmland birds and resource protection in the UK: Cross-cutting solutions for multi-functional farming? *Biological Conservation*, *129*(4), 530–542.

Burger, L. D., Burger, L. W., & Faaborg, J. (1994). Effects of Prairie Fragmentation on Predation on Artificial Nests. *The Journal of Wildlife Management*, *58*(2), 249–254.

Casanova, M. T., & Brock, M. A. (2000). How do depth, duration and frequency of flooding influence the establishment of wetland plant communities? *Plant Ecology*, *147*(2), 237–250.

Castelli, R. M., Chambers, J. C., & Tausch, R. J. (2000). Soil-Plant relations Along a Soil-Water Gradient in Great Basin Riparian Meadows. *Wetlands*, *20*(2), 251–266.

CBS. (2015). Weidevogels in duikvlucht. Retrieved from https://www.cbs.nl/nl-nl/nieuws/2015/32/weidevogels-in-duikvlucht on 07-10-2019

CBS. (2017). Nederlandse landbouwproductie 1950-2015. Retrieved from https://www.cbs.nl/nl-nl/nieuws/2017/05/nederlandse-landbouwproductie-1950-2015 on 06-10-2019

Davis, C. A., Austin, J. E., & Buhl, D. A. (2006). Factors influencing soil invertebrate communities in riparian grasslands of the central platte river floodplain. *Wetlands*, *26*(2), 438–454.

Davis, S. K. (2005). Nest-Site Selection Patterns and the Influence of Vegetation on Nest Survival of Mixed-Grass Prairie Passerines. *The Condor*, *107*(3), 605–616.

De Jong, H., & Van Zanden, J. L. (2014). Debates on industrialisation and economic growth in the Netherlands. *Tijdschrift Voor Sociale En Economische Geschiedenis*, *11*(2), 85–109.
De Snoo, G. R., Naus, N., Verhulst, J., van Ruijven, J., & Schaffers, A. P. (2012). Long-term changes in plant diversity of grasslands under agricultural and conservation management. *Applied Vegetation Science*, *15*(3), 299–306.
De Vries, D. M. (1929). Het Plantendek van Krimpenerwaard III: Over de samenstelling van het crempensch Molinietum coruleae en Agrostidetum caninae. Een phytostatische bijdrage tot de associatie-wetenschap. *Nederlands Kruidkundig Archief*, *39*(2), 145–403.
Dekkers, G. (2002). De vroegste geschiedenis van Nederland. *Historisch Nieuwsblad*, *9*.
Dennis, P., Young, M. R., & Gordon, I. J. (1998). Distribution and abundance of small insects and arachnids in relation to structural heterogeneity of grazed, indigenous grasslands. *Ecological Entomology*, *23*, 253–264.
Dieter, D., Herzog, C., & Hupfer, M. (2015). Effects of drying on phosphorus uptake in re-flooded lake sediments. *Environmental Science and Pollution Research*, *22*(21), 17065–17081.
Dinno, A. (2017). Dunn's test of multiple comparisons using rank sums.
Donald, P. F., Green, R. E., & Heath, M. F. (2001). Agricultural intensification and the collapse of Europe’s farmland bird populations. *Proceedings of the Royal Society B: Biological Sciences*, *268*(1462), 25–29.
Dwire, K. A., Kauffman, J. B., & Baham, J. E. (2006). Plant species distribution in relation to water-table depth and soil redox potential in montane riparian meadows. *Wetlands*, *26*(1), 131–146.
Ejankowski, W., & Solis, M. (2014). Response of Hornwort (Ceratophyllum demersum L.) to water level drawdown in a turbid water reservoir. *Applied Ecology and Environmental Research*, *13*(1), 219–228.
Ernst, L. M., Tscharntke, T., & Batáry, P. (2017). Grassland management in agricultural vs. forested landscapes drives butterfly and bird diversity. *Biological Conservation*, *216*(September), 51–59.
European Commission. (2019). Biodiversity strategy. Retrieved from https://ec.europa.eu/environment/nature/biodiversity/strategy/index_en.htm on 07-10-2019
Feminella, J. W., & Resh, V. H. (1989). Submerged macrophytes and grazing crayfish: an experimental study of herbivory in a California freshwater marsh. *Holarctic Ecology*, *12*, 1–8.
Feng, H. (1998). *Agricultural Development in the Netherlands*.
Flohre, A., Fischer, C., Aavik, T., Bengtsson, J., Berendse, F., Bommarco, R., … Tscharntke, T. (2011). Agricultural intensification and biodiversity partitioning in European landscapes comparing plants, carabids, and birds. *Ecological Applications*, *21*(5), 1772–1781.
Foster, B. L., Kindscher, K., Houseman, G. R., & Murphy, C. A. (2009). Effects of hay management and native species sowing on grassland community structure, biomass, and restoration. *Ecological Applications*, *19*(7), 1884–1896.
Fraixedas, S., Lindén, A., Meller, K., Lindström, Å., Keišs, O., Kålås, J. A., … Lehikoinen, A. (2017). Substantial decline of Northern European peatland bird populations: Consequences of drainage. *Biological Conservation*, *214*(March), 223–232.
Fredrickson, L. H., & Reid, F. A. (1988). Nutritional Values of Waterfowl Foods. In *Waterfowl Management Handbook*.
Gerritsen, G., & Roodhart, J. (2018). Black-Tailed Godwit flying high in Utrecht. Retrieved from https://www.birdlife.org/europe-and-central-asia/news/black-tailed-godwit-flying-high-utrecht on 12-11-2019
Görn, S., Schulze, F., & Fischer, K. (2015). Effects of fen management on bird communities in north-eastern Germany. *Journal of Ornithology*, *156*(1), 287–296.
Green, R. E., Hirons, G. J. M., & Cresswell, B. H. (1990). Foraging Habitats of Female Common Snipe Gallinago gallinago During the Incubation Period. *Journal of Applied Ecology*, *27*(1), 325–335.
Hammersmark, C. T., Rains, M. C., Wickland, A. C., & Mount, J. F. (2009). Vegetation and water-table relationships in a hydrologically restored riparian meadow. *Wetlands*, *29*(3), 785–797.
Helzer, C.J. (1996). The Effects of Wet Meadow Fragmentation on Grassland Birds. *Dissertations & Thesis in Natural Resources*, *193*.

Helzer, Christopher J., & Jelinski, D. E. (1999). The relative importance of patch area and perimeter-area ratio to grassland breeding birds. *Ecological Applications*, *9*(4), 1448–1458.
Hennekes, S. M., Smits, N. A. C., & Schaminee, J. H. J. (2010). SynBioSys Nederland versie 3. Alterra, Wageningen UR.
Henszey, R. J., Pfeiffer, K., & Keough, J. R. (2004). Linking surface- and ground-water levels to riparian grassland species along the Platte River in central Nebraska, USA. *Wetlands*, *24*(3), 665–687.
Hoek, D. van der, & Kemmers, R. H. (1998). Invloed van 10 jaar vernatting op de regeneratieprocessen in de bodem van De Veenkampen. In M. J. M. Oomes & H. Korevaar (Eds.), *Herstel van natte, soortenrijke graslanden.*
Ihse, M. (1995). Swedish agricultural landscapes - patterns and changes during the last 50 years, studied by aerial photos. *Landscape and Urban Planning*, *31*(1–3), 21–37.
Kieckbusch, J. J., & Schrautzer, J. (2007). Nitrogen and phosphorus dynamics of a re-wetted shallow-flooded peatland. *Science of the Total Environment*, *380*(1–3), 3–12.
Kleijn, D., Kohler, F., Báldi, A., Batáry, P., Concepción, E. D., Clough, Y., … Verhulst, J. (2012). On the relationship between farmland biodiversity and land-use intensity in Europe. *Proceedings of the Royal Society B: Biological Sciences*, *276*(1658), 903–909.
Kleijn, David, Berendse, F., Smit, R., Gilissen, N., Smit, J., & Groeneveld, R. (2004). Effectiveness of Agri-Environment Schemes in Different Agricultural Landscapes in the Netherlands. *Conservation Biology*, *18*(3), 775–7868.
Kleinjan, S. (2019). Personal Comment.
Klimešová, J. (1994). The effects of timing and duration of floods on growth of yound plants of Phalaris arundinacea L. and Urtica dioica L.: an experimental study. *Aquatic Botany*, *48*(1), 21–29.
Knibbe, M. T. (2000). Feed, Fertilizer, and Agricultural Productivity in the Netherlands, 1880-1930. *Agricultural History*, *74*(1), 39–57.
KNMI. (2018). Neerslagtekort in Nederland in 2018. Retrieved from https://www.knmi.nl/nederland-nu/klimatologie/geografische-overzichten/historisch-neerslagtekort on 12-11-2019
Koch, M. S., Mendelssohn, I. A., & McKee, K. L. (1990). Mechanism for the hydrogen sulfide-induced growth limitation in wetland macrophytes. *Limnology and Oceanography*, *35*(2), 399–408.
Lamers, L., Lucassen, E., Smolders, F., & Roelofs, J. (2005). Fosfaat als adder onder het gras bij “nieuwe natte natuur.” *H2O: Tijdschrift Voor Watervoorziening En Afvalwaterbehandeling*, *38*(17), 28–30.
Lamers, L. P. M., Govers, L. L., Janssen, I. C. J. M., Geurts, J. J. M., Van der Welle, M. E. W., Van Katwijk, M. M., … Smolders, A. J. P. (2013). Sulfide as a soil phytotoxin-a review. *Frontiers in Plant Science*, *4,* 1–14.
Lamers, L. P. M., Tomassen, H. B. M., & Roelofs, J. G. M. (1998). Sulfate-induced eutrophication and phytotoxicity in freshwater wetlands. *Environmental Science and Technology*, *32*(2), 199–205.
Laurance, W. F., Nascimento, H. E. M., Laurance, S. G., Andrade, A., Ewers, R. M., Harms, K. E., … Ribeiro, J. E. (2007). Habitat fragmentation, variable edge effects, and the landscape-divergence hypothesis. *PLoS ONE*, *2*(10).
Lemmers, P., Crombaghs, B. H. J. M., & Leuven, R. S. E. W. (2018). Invasieve exotische kreeften in het beheergebied van waterschap Rivierenland, 91.
Leng, R. A., Stambolie, J. H., & Bell, R. (1995). Duckweed - a potential high-protein feed source for domestic animals and fish. *Livestock Research for Rural Development*, *7*(1).
Luoto, M., Rekolainen, S., Aakkula, J., & Pykälä, J. (2003). Loss of Plant Species Richness and Habitat Connectivity in Grasslands Associated with Agricultural Change in Finland. *Ambio*, *32*(7), 447–452.
Mauchamp, A., Chauvelon, P., & Grillas, P. (2002). Restoration of floodplain wetlands: Opening polders along a coastal river in Mediterranean France, Vistre marshes. *Ecological Engineering*, *18*(5), 619–632.
Milsom, T. P., Langton, S. D., Parkin, W. K., Peel, S., Bishop, J. D., Hart, J. D., & Moore, N. P. (2000). Habitat models of bird species’ distribution: An aid to the management of coastal grazing

marshes. *Journal of Applied Ecology*, *37*(5), 706–727.
Mischenko, A., Sukhanova, O., & Zöckler, C. (2014). Rewetting the Vinogradovo Floodplain, Moscow region, Russia: A project in support of wader populations. *Wader Study Group Bulletin*, *121*(2), 77–80.
Morris, M. G. (2000). The effects of structure and its dynamics on the ecology and conservation of arthropods in British grasslands. *Biological Conservation*, *95*, 129–142.
NDFF. (2007). NDFF Verspreidingsatlas. Retrieved from https://www.verspreidingsatlas.nl/2620# on 05-11-2019
Olde Venterink, H., Davidsson, T. E., Kiehl, K., & Leonardson, L. (2002). Impact of drying and rewetting on carbon cycling in a northern fen. *Plant and Soil*, *243*, 119–130.
Olivier, R. S., Aranda, R., Godoi, M. N., & Gracioll, G. (2014). Effects of environmental heterogeneity on the composition of insect trophic guilds. *Applied Ecology and Environmental Research*, *12*(1), 209–220.
Oomes, M. J. M., Olff, H., & Altena, H. J. (1996). Effects of Vegetation Management and Raising the Water Table on Nutrient Dynamics and Vegetation Change in a Wet Grassland. *The Journal of Applied Ecology*, *33*(3), 576.
Pannek, A., Duprè, C., Gowing, D. J. G., Stevens, C. J., & Diekmann, M. (2015). Spatial gradient in nitrogen deposition affects plant species frequency in acidic grasslands. *Oecologia*, *177*(1), 39–51.
Peintinger, M., Prati, D., & Winkler, E. (2007). Water level fluctuations and dynamics of amphibious plants at Lake Constance: Long-term study and simulation. *Perspectives in Plant Ecology, Evolution and Systematics*, *8*(4), 179–196.
Perry, L. G., Galatowitsch, S. M., & Rosen, C. J. (2004). Competitive control of invasive vegetation: a native wetland sedge suppresses Phalaris arundinacea in carbon-enrichted soil. *Journal of Applied Ecology*, *41*, 151–162.
Plieninger, T., Levers, C., Mantel, M., Costa, A., Schaich, H., & Kuemmerle, T. (2015). Patterns and drivers of scattered tree loss in agricultural landscapes: Orchard meadows in Germany (1968-2009). *PLoS ONE*, *10*(5), 1–19.
Primdahl, J., Peco, B., Schramek, J., Andersen, E., & Oñate, J. J. (2003). Environmental effects of agri-environmental schemes in Western Europe. *Journal of Environmental Management*, *67*(2), 129–138.
Provincie Groningen. (2014). *Toestand Natuur en Landschap 2014 in de provincie Groningen*.
QGIS Development Team. (2018). QGIS Geographic Information System. Open Source Geospatial Foundation Project.
Queiroz, C., Beilin, R., Folke, C., & Lindborg, R. (2014). Farmland abandonment: Threat or opportunity for biodiversity conservation? A global review. *Frontiers in Ecology and the Environment*, *12*(5), 288–296.
R Core Team (2018). R: A language and environment for statistical computing.
Reidsma, P., Tekelenburg, T., Van Den Berg, M., & Alkemade, R. (2006). Impacts of land-use change on biodiversity: An assessment of agricultural biodiversity in the European Union. *Agriculture, Ecosystems and Environment*, *114*(1), 86–102.
Rhymer, C. M., Robinson, R. A., Smart, J., & Whittingham, M. J. (2010). Can ecosystem services be integrated with conservation? A case study of breeding waders on grassland. *Ibis*, *152*(4), 698–712.
Riggins, J. J., Davis, C. A., & Hoback, W. W. (2009). Biodiversity of belowground invertebrates as an indicator of wet meadow restoration success (Platte River, Nebraska). *Restoration Ecology*, *17*(4), 495–505.
Rodríguez, C. F., Bécares, E., Fernández-Aláez, M., & Fernández-Aláez, C. (2005). Loss of diversity and degradation of wetlands as a result of introducing exotic crayfish. *Biological Invasions*, *7*, 75–85.
Roelofs, J. G. M. (1991). Inlet of alkaline river water into peaty lowlands: effects on water quality and Stratiotes aloides L. stands. *Aquatic Botany*, *39*(3–4), 267–293.
Romero-Alcaraz, E., & Ávila, J. M. (2000). Landscape heterogeneity in relation to variations in

epigaeic beetle diversity of a Mediterranean ecosystem. Implications for conservation. *Biodiversity and Conservation*, *9*(7), 985–1005.
Scheffer, M., Hosper, S. H., Meijer, M. L., Moss, B., & Jeppesen, E. (1993). Alternative Equilibria in Shallow Lakes. *Trends in Ecology and Evolution*, *8*(8), 275–279.
Schekkerman, H., Teunissen, W., & Oosterveld, E. (2009). Mortality of Black-tailed Godwit Limosa limosa and Northern Lapwing Vanellus vanellus chicks in wet grasslands: Influence of predation and agriculture. *Journal of Ornithology*, *150*(1), 133–145.
Schep, S. A., & Verbeek, S. K. (2018). *Ecologische Sleutelfactoren*. *Landschap* (Vol. 24).
Smolders, A. J. P., Van de Riet, B. P., Van Diggelen, J. M. H., Van Dijk, G., Geurts, J. J. M., & Lamers, L. P. M. (2019). De toekomst van ons veenweidelandschap. Over vernatten, optoppen en veenmosteelt. *Landschap*, *3*, 132–141.
Stanton, R. L., Morrissey, C. A., & Clark, R. G. (2018). Analysis of trends and agricultural drivers of farmland bird declines in North America: A review. *Agriculture, Ecosystems and Environment*, *254*, 244–254.
Steinman, A. D., Ogdahl, M. E., Weinert, M., & Uzarski, D. G. (2014). Influence of water-level fluctuation duration and magnitude on sediment-water nutrient exchange in coastal wetlands. *Aquatic Ecology*, *48*(2), 143–159.
Terlouw, R. J. S. (2015). *Graslanden Inventarisaties t.b.v. Evaluatie Pilot Natuurbeheer Krimpenerwaard [polder de Nesse en Kattendijksblok]*. Oudekerk aan den IJssel.
Teunissen, W., Schekkerman, H., & Willems, F. (2005). *Predatie bij weidevogels*. *Alterra*.
Timmermann, T., Margóczi, K., Takács, G., & Vegelin, K. (2006). Restoration of peat-forming vegetation by rewetting species-poor fen grasslands. *Applied Vegetation Science*, *9*(2), 241–250.
Toogood, S. E., & Joyce, C. B. (2009). Effects of raised water levels on wet grassland plant communities. *Applied Vegetation Science*, *12*(3), 283–294.
Uchida, K., & Ushimaru, A. (2014). Biodiversity declines due to abandonment and intensification of agricultural lands: patterns and mechanisms. *Ecological Monographs*, *84*(4), 637–658.
Valipour, R., Boegman, L., Bouffard, D., & Rao, Y. R. (2017). Sediment resuspension mechanisms and their contributions to high-turbidity events in a large lake. *Limnology and Oceanography*, *62*(3), 1045–1065.
Van Andel, J., & Aronson, J. (2006). *Restoration Ecology: The New Frontier*. *Ecological Management and Restoration* (Vol. 15). Oxford: Blackwell Science Ltd.
Van der Wal, J. E. M., Dorenbosch, M., Immers, A. K., Vidal Forteza, C., Geurts, J. J. M., Peeters, E. T. H. M., … Bakker, E. S. (2013). Invasive Crayfish Threaten the Development of Submerged Macrophytes in Lake Restoration. *PLoS ONE*, *8*(10), 1–11.
Van der Weijden, A. G. G., & Guldemond, J. A. (2006). Wormenland en vliegjesland: bemesting in relatie tot voedsel voor de grutto.
Van der Winden, J., Courbois, M., Van Horssen, P., Koenders, W., Kanters, S., & Poot, M. (2018). *Effect natuurmaatregelen in Polder Berkenwoude en de Nesse*.
Van der Winden, J., Kanters, S., Poot, M., & Van Horssen, P. (2017). *Natuurmaatregelen in de Krimpenerwaard positief voor vogels? Effecten in het eerste jaar na aanleg in Polder*.
Van Dijk, J. (2008). Vernatting in het westelijk veen- weidegebied. *Landschap*, *25*(1), 5–15.
Van Donk, S., Courbois, M., Koenders, W., & Van der Winden, J. (2019). *Effect natuurmaatregelen in Polder Berkenwoude en de Nesse. Jaarrapport 2019: veranderingen in biodiversiteit vogels, insecten en regenwormen.*
Van Gerven L.P.A., Hendriks R.F.A., Harmsen J., Beumer V., & Bogaart P.W. (2011). *Nalevering van fosfor naar het oppervlaktewater vanuit de waterbodem: Metingen in een veengebied in de Krimpenerwaard*. *Alterra-rapport 2217*.
Van Turnhout, C. A. M., Foppen, R. P. B., Leuven, R. S. E. W., Siepel, H., & Esselink, H. (2007). Scale-dependent homogenization: Changes in breeding bird diversity in the Netherlands over a 25-year period. *Biological Conservation*, *134*(4), 505–516.
Van Veen, M.P., ten Brink, B.J.E., Braat, L.C. & Melman, T. C. P. (2008). *Halting biodiversity loss in the Netherlands*. *PBL (Netherlands Environmental Assessment Agency).*

Vandenberghe, C., Prior, G., Littlewood, N. A., Brooker, R., & Pakeman, R. (2009). Influence of livestock grazing on meadow pipit foraging behaviour in upland grassland. *Basic and Applied Ecology*, *10*(7), 662–670.

Verhoeven, J., Barendregt, A., & Van De Riet, B. (2010). Kansen voor natuur veenweidegebied. *Landschap*, *27*(3), 157–165.

Verhulst, J., Kleijn, D., & Berendse, F. (2007). Direct and indirect effects of the most widely implemented Dutch agri-environment schemes on breeding waders. *Journal of Applied Ecology*, *44*(1), 70–80.

Vermaat, J. E., Goosen, H., & Omtzigt, N. (2007). Do biodiversity patterns in Dutch wetland complexes relate to variation in urbanisation, intensity of agricultural land use or fragmentation? *Biodiversity and Conservation*, *16*(12), 3585–3595.

Visser, T., Melman, D., Buij, R., & Schotman, A. (2017). *Greppel plas-dras voor weidevogels Betekenis als habitatonderdeel voor weidevogelkuikens*.

Vonk, J. A., Rombouts, T., Schoorl, J. C., Serne, P., Westerveld, J. W., Cornelissen, P., & van der Geest, H. G. (2017). Impact of water drawdown and rewetting on sediment nutrient-dynamics in a constructed delta-lake system (Oostvaardersplassen, The Netherlands): A mesocosm study. *Ecological Engineering*, *108*, 396–405.

Warren, K. A., & Anderson, J. T. (2005). Grassland songbird nest-site selection and response to mowing in West Virginia. *Wildlife Society Bulletin*, *33*(1), 285–292.

Wereld Natuur Fonds. (2015). *Living Planet Report. Natuur in Nederland*. Zeist.

Westerink, J., Plomp, M., Ottburg, F., Zanen, M., & Schrijver, R. (2018). *Boeren voor Natuur: de ultieme natuurinclusieve landbouw?*

Wickham, H. (2007). Reshaping Data with the reshape Package. *Journal of Statistical Software*, *21*(12), 1–20.

Wickham, H. (2016). ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag New York.

Yamanaka, S., Akasaka, T., Yabuhara, Y., & Nakamura, F. (2017). Influence of farmland abandonment on the species composition of wetland ground beetles in Kushiro, Japan. *Agriculture, Ecosystems and Environment*, *249*, 31–37.

Zak, D., Gelbrecht, J., & Steinberg, C. E. W. (2004). Phosphorus retention at the redox interface of peatlands adjacent to surface waters in Northeast Germany. *Biogeochemistry*, *70*(3), 357–368.

Zak, Dominik, & Gelbrecht, J. (2007). The mobilisation of phosphorus, organic carbon and ammonium in the initial stage of fen rewetting (a case study from NE Germany). *Biogeochemistry*, *85*(2), 141–151.

Zmihorski, M., Pärt, T., Gustafson, T., & Berg, Å. (2016). Effects of water level and grassland management on alpha and beta diversity of birds in restored wetlands. *Journal of Applied Ecology*, *53*(2), 587–595.

APPENDIX A

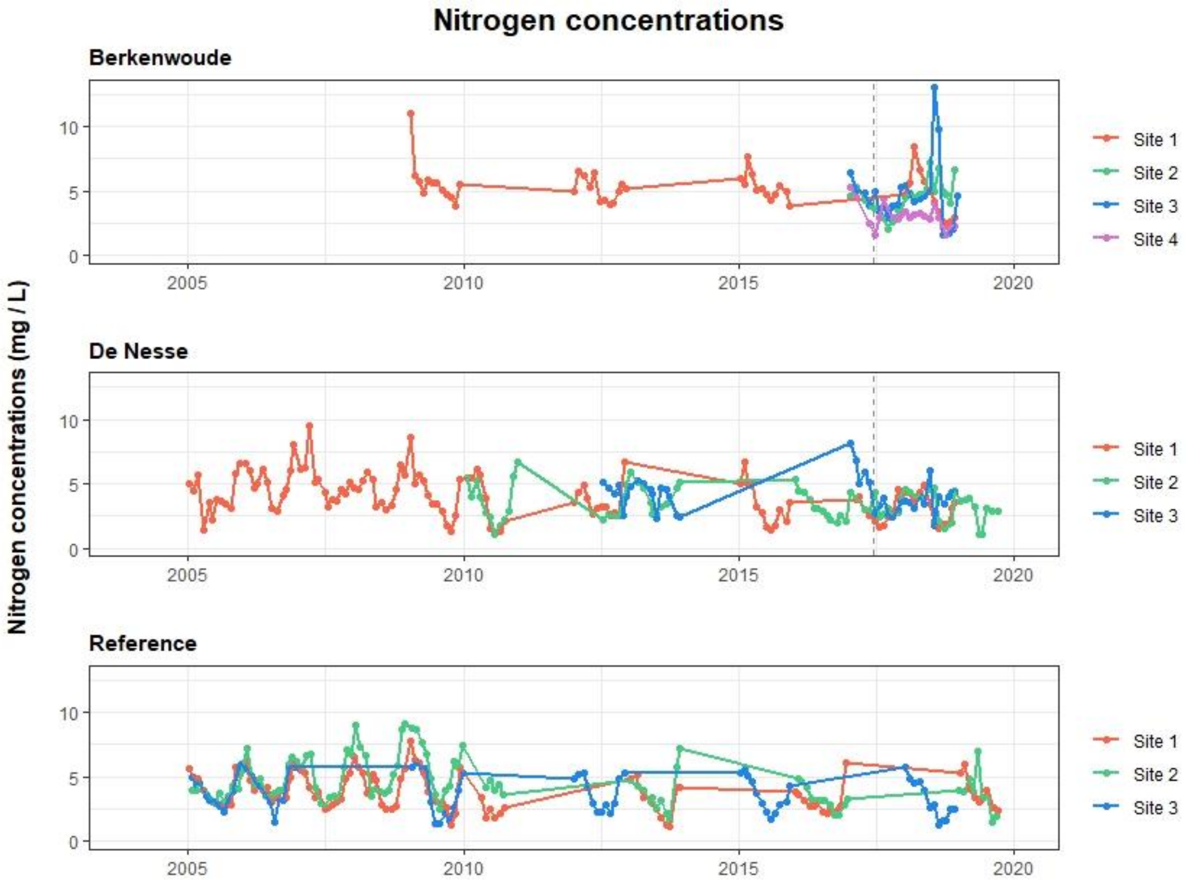

Figure 1 Total nitrogen concentrations (mg L^{-1}) in Berkenwoude, De Nesse and the reference area since the start of the first monitoring in 2005. The dashed line gives the water level increase in Berkenwoude and De Nesse. In the reference area, water levels were not increased.

Table 1 Z-values and p-values of Dunn's post-hoc test (Kruskal-Wallis test outcome: $X^2 = 14.33$, $p < 0.01$) for total nitrogen concentrations between the different polders. Significance levels are: * = p < 0.05, ** = p < 0.01, *** = p < 0.001.

	De Nesse	Reference
Berkenwoude	Z = 3.76, p < 0.01***	Z = 2.86, p < 0.01**
De Nesse		Z = 1.24, p = 0.11

Table 2 Z-values and p-values of Dunn's post-hoc test (Kruskal-Wallis test outcome: $X^2 = 32.60$, $p < 0.01$) for total nitrogen concentrations during the monitoring period at the different monitoring sites in Berkenwoude. Significance levels are: * = p < 0.05, ** = p < 0.01, *** = p < 0.001.

BERKENWOUDE			
	Site 2	**Site 3**	**Site 4**
Site 1	Z = 2.06, p = 0.02*	Z = 2.00, p = 0.02*	Z = 5.07, p < 0.01***
Site 2		Z = 0.05, p = 0.48	Z = 3.18, p < 0.01***
Site 3			Z = 3.22, p < 0.01***

Table 3 W-values and p-values of Wilcoxon's rank-sum test for the differences in nitrogen concentration before and after water level increase (June 2017). Note that in the reference area, water levels were not increased. Significance levels are: * = p < 0.05, ** = p < 0.01, *** = p < 0.001.

	Berkenwoude	De Nesse	Reference area
Site 1	W = 280, p = 0.13	W = 1288, p < 0.01**	W = 391, p = 0.87
Site 2	W = 41, p = 0.84	W = 701.5, p = 0.42	W = 499, p = 0.20
Site 3	W = 52.5, p = 0.26	W = 321.5, p < 0.01**	W = 461, p = 0.13
Site 4	W = 43, p = 0.18		

Table 4 Z-values and p-values of Dunn's post-hoc test (Kruskal-Wallis test outcome: $X^2 = 8.31$, p = 0.02) for total nitrogen concentrations during the monitoring period at the different monitoring sites in De Nesse. Significance levels are: * = p < 0.05, ** = p < 0.01, *** = p < 0.001.

DE NESSE		
	Site 2	**Site 3**
Site 1	Z = 2.50, p < 0.01**	Z = 0.55, p = 0.29
Site 2		Z = 2.44, p < 0.01**

Table 5 Z-values and p-values of Dunn's post-hoc test (Kruskal-Wallis test outcome: $X^2 = 12.18$, p < 0.01) for total nitrogen concentrations during the monitoring period at the different monitoring sites in the reference area. Significance levels are: * = p < 0.05, ** = p < 0.01, *** = p < 0.001.

REFERENCE AREA		
	Site 2	**Site 3**
Site 1	Z = 3.16 p < 0.01***	Z = 0.12, p = 0.45
Site 2		Z = 2.78, p < 0.01**

Table 6 Z-values and p-values of Dunn's post-hoc test (Kruskal-Wallis test outcome: $X^2 = 117.86$, p < 0.01) for phosphate concentrations between the different polders. Significance levels are: * = p < 0.05, ** = p < 0.01, *** = p < 0.001.

	De Nesse	Reference area
Berkenwoude	Z = 6.04, p < 0.01***	Z = 10.67, p < 0.01***
De Nesse		Z = 5.61, p < 0.01***

Table 7 Z-values and p-values of Dunn's post-hoc test (Kruskal-Wallis test outcome: $X^2 = 40.81$, p < 0.01) for phosphate concentrations during the monitoring period at the different monitoring sites in Berkenwoude. Significance levels are: * = p < 0.05, ** = p < 0.01, *** = p < 0.001.

BERKENWOUDE			
	Site 2	**Site 3**	**Site 4**
Site 1	Z = 5.98, p < 0.01***	Z = 0.58, p = 0.28	Z = 3.19, p < 0.01***
Site 2		Z = 4.63, p < 0.01***	Z = 2.34, p < 0.01**
Site 3			Z = 2.24, p = 0.01*

Table 8 W-values and p-values of Wilcoxon's rank-sum test for the differences in phosphate concentration before and after water level increase (June 2017). Note that in the reference area, water levels were not increased. Significance levels are: * = p < 0.05, ** = p < 0.01, *** = p < 0.001.

	Berkenwoude	De Nesse	Reference area
Site 1	W = 276, p = 0.16	W = 1066, p = 0.79	W = 760, p < 0.01**
Site 2	W = 48, p = 0.44	W = 915.5, p < 0.01**	W = 623, p = 0.06
Site 3	W = 59, p = 0.10	W = 134, p = 0.03*	W = 731 p < 0.01***
Site 4	W = 24, p = 0.70		

Table 9 Z-values and p-values of Dunn's post-hoc test (Kruskal-Wallis test outcome: X^2 = 56.15, p < 0.01) for phosphate concentrations during the monitoring period at the different monitoring sites in De Nesse. Significance levels are: * = p < 0.05, ** = p < 0.01, *** = p < 0.001.

	DE NESSE	
	Site 2	**Site 3**
Site 1	Z = 7.35, p < 0.01***	Z = 0.85, p = 0.20
Site 2		Z = 4.78, p < 0.01***

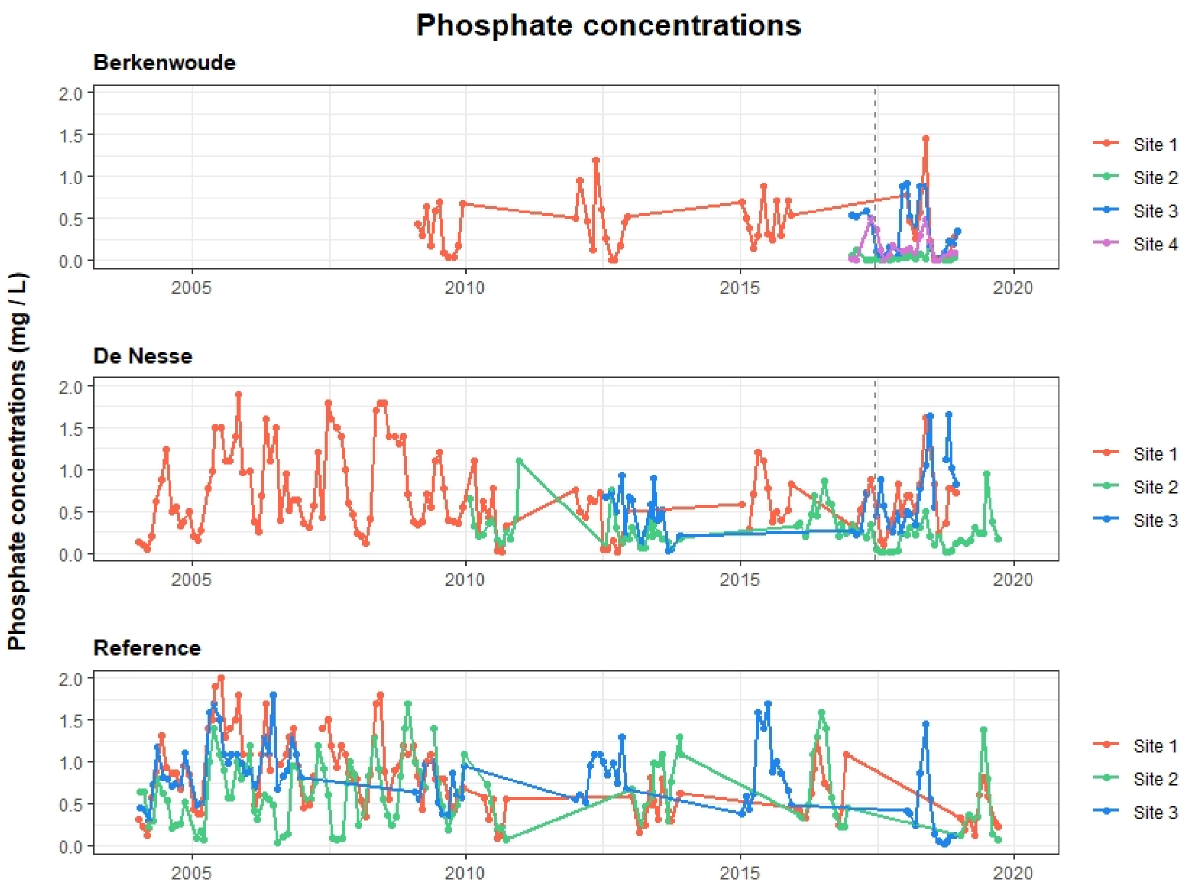

Figure 2 Phosphate concentrations (mg L^{-1}) in Berkenwoude, De Nesse and the reference area since 2004. The dashed line gives the water level increase in Berkenwoude and De Nesse. In the reference area, water levels were not increased.

Table 10 Z-values and p-values of Dunn's post-hoc test (Kruskal-Wallis test outcome: X^2= 11.30, $p < 0.01$) for phosphate concentrations during the monitoring period at the different monitoring sites in the reference area. Significance levels are: * = p < 0.05, ** = p < 0.01, *** = p < 0.001.

REFERENCE AREA		
	Site 2	**Site 3**
Site 1	Z = 2.76, p < 0.01**	Z = 0.43, p = 0.33
Site 2		Z = 2.98, p < 0.01**

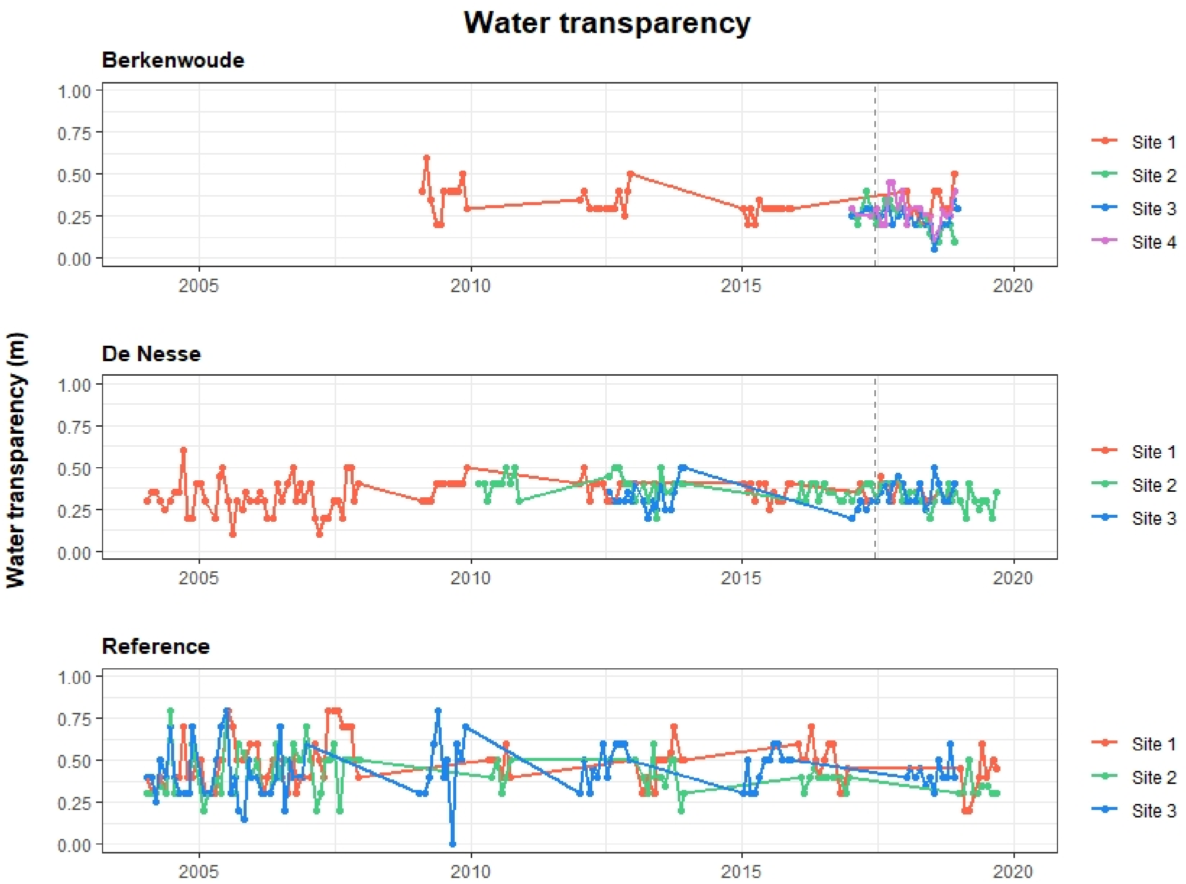

Figure 3 Water transparency (m) in Berkenwoude, De Nesse and the reference area since 2004. The dashed line indicates the water level increase in Berkenwoude and De Nesse. In the reference area, water levels have not been increased.

Table 11 Z-values and p-values of Dunn's post-hoc test (Kruskal-Wallis test outcome: X^2 = 140.81, $p < 0.01$) for water transparency between the different polders. Significance levels are: * = p < 0.05, ** = p < 0.01, *** = p < 0.001.

	De Nesse	**Reference area**
Berkenwoude	Z = 5.18, p < 0.01***	Z = 11.39, p < 0.01***
De Nesse		Z = 7.40, p < 0.01***

Table 12 Z-values and p-values of Dunn's post-hoc test (Kruskal-Wallis test outcome: X^2 = 24.13, $p < 0.01$) for water transparency during the monitoring period at the different monitoring sites in Berkenwoude. Significance levels are: * = p < 0.05, ** = p < 0.01, *** = p < 0.001.

BERKENWOUDE			
	Site 2	**Site 3**	**Site 4**
Site 1	Z = 5.98, p < 0.01***	Z = 0.58, p = 0.28	Z = 3.19, p < 0.01***
Site 2		Z = 4.63, p < 0.01***	Z = 2.34, p < 0.01**
Site 3			Z = 2.24, p = 0.01*

Table 13 W-values and p-values of Wilcoxon's rank-sum test for the differences in water transparency before and after water level increase (June 2017). Note that in the reference area, water levels were not increased. Significance levels are: * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$.

	Berkenwoude	De Nesse	Reference area
Site 1	W = 238, p = 0.48	W = 711.5, p = 0.58	W = 477.5, p = 0.07
Site 2	W = 57, p = 0.12	W = 899, p < 0.01**	W = 482.5, p = 0.04*
Site 3	W = 55, p = 0.16	W = 138, p = 0.04*	W = 442.5, p = 0.77
Site 4	W = 26, p = 0.84		

Table 14 Z-values and p-values of Dunn's post-hoc test (Kruskal-Wallis test outcome: $X^2 = 11.30$, $p < 0.01$) for water transparency during the monitoring period at the different monitoring sites in the reference area. Significance levels are: * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$.

REFERENCE AREA		
	Site 2	**Site 3**
Site 1	Z = 3.45, p < 0.01***	Z = 1.84, p = 0.03*
Site 2		Z = 1.57, p = 0.06

APPENDIX B

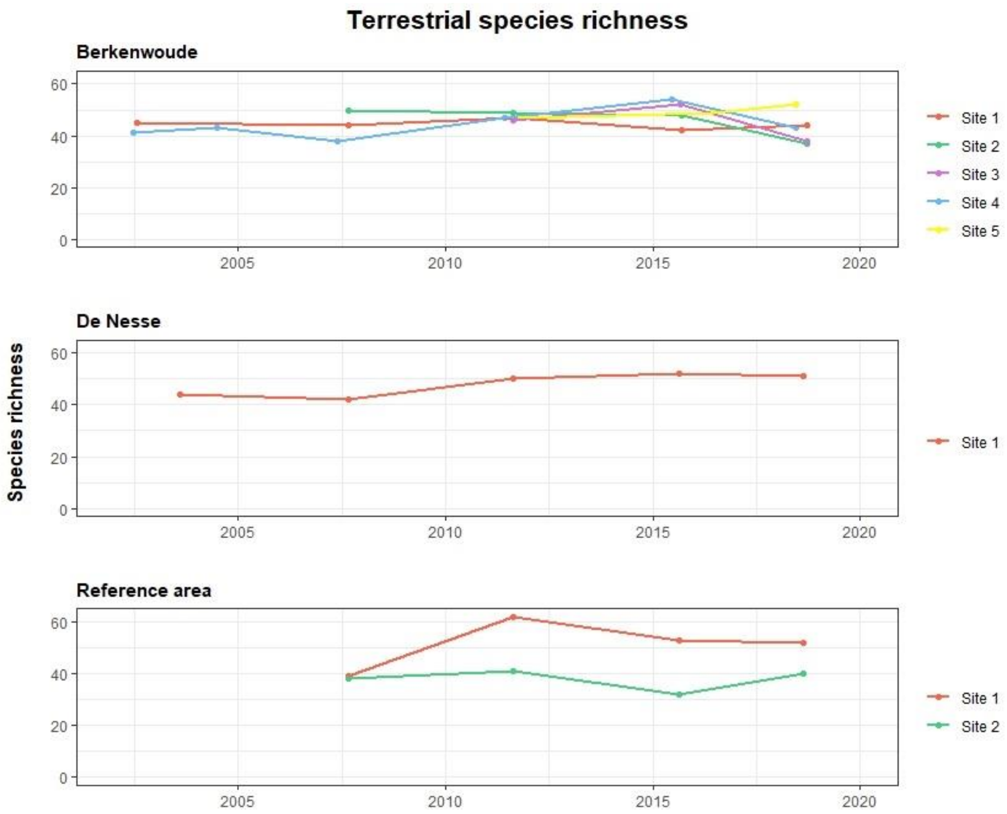

Figure 1 Species richness over time at the terrestrial monitoring sites in Berkenwoude, De Nesse and the reference area. Site-specific colours correspond with the numbers of the monitoring sites as given in the methods section.

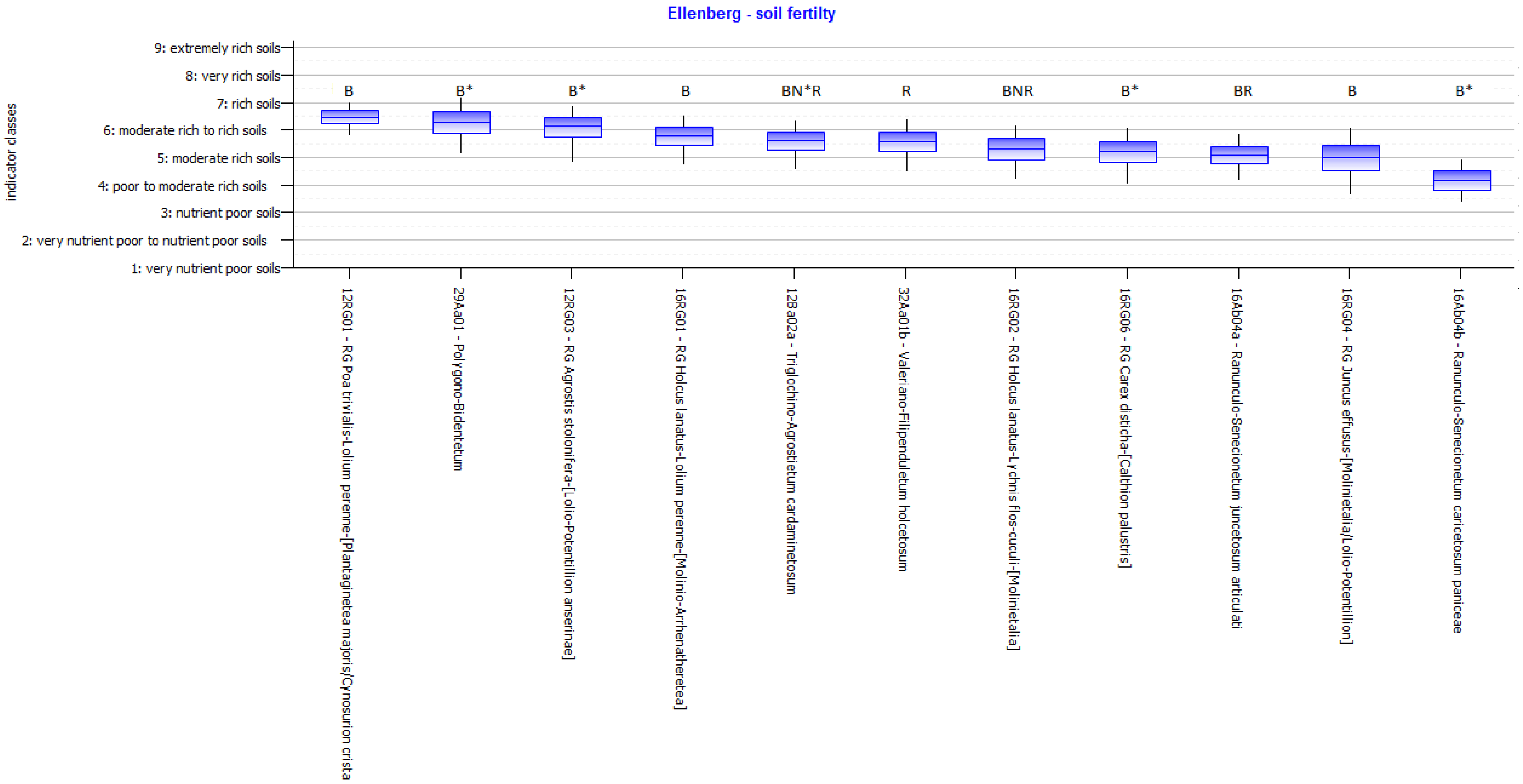

Figure 2 Ellenberg fertility classes of the vegetation types that were present at the terrestrial monitoring sites in the three areas – Berkenwoude (B), De Nesse (N), reference area (R). Vegetation types that were present after the water level increase in June 2017 in Berkenwoude and De Nesse are indicated with an asterisk. As water levels were not increased in the reference area, vegetation types that were observed after June 2017 in the reference area are not indicated with an asterisk. Adapted from: Hennekes et al., 2010.

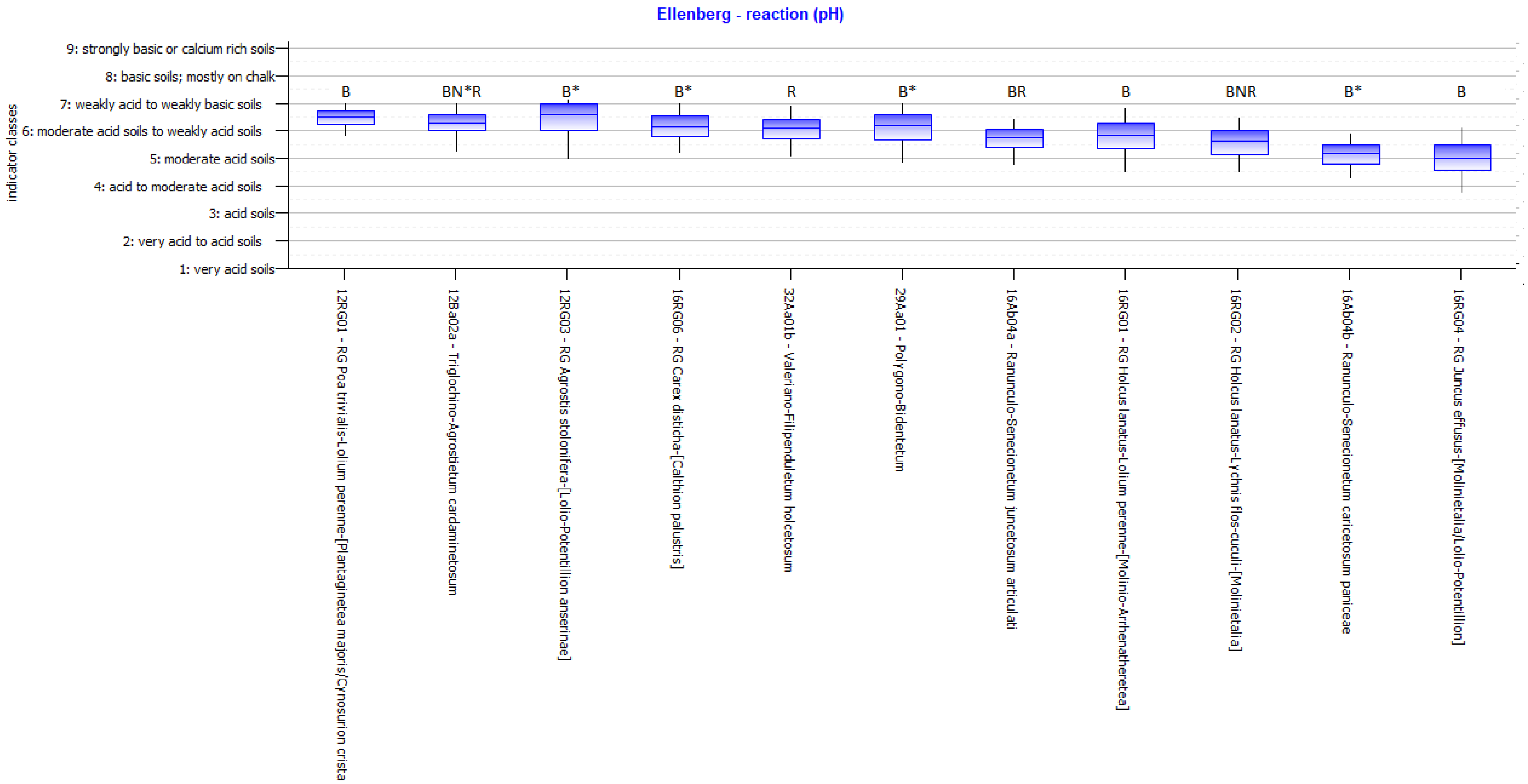

Figure 3 Ellenberg acidity classes of the vegetation types that were present at the terrestrial monitoring sites in the three areas – Berkenwoude (B), De Nesse (N), reference area (R). Vegetation types that were present after the water level increase in June 2017 in Berkenwoude and De Nesse are indicated with an asterisk. As water levels were not increased in the reference area, vegetation types that were observed after June 2017 in the reference area are not indicated with an asterisk. Adapted from: Hennekes et al., 2010.

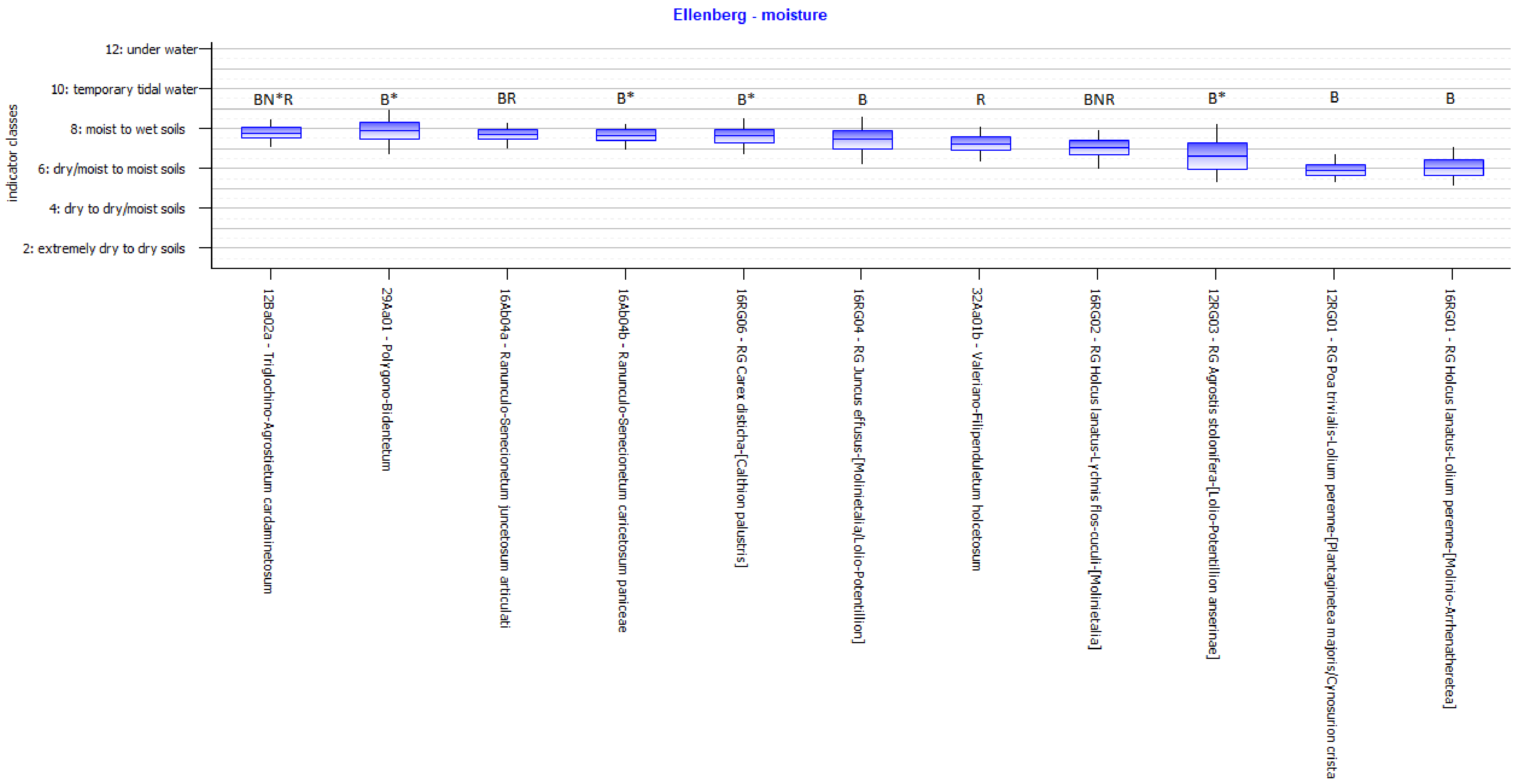

Figure 4 Ellenberg moisture classes of the vegetation types that were present at the terrestrial monitoring sites in the three areas – Berkenwoude (B), De Nesse (N), reference area (R). Vegetation types that were present after the water level increase in June 2017 in Berkenwoude and De Nesse are indicated with an asterisk. As water levels were not increased in the reference area, vegetation types that were observed after June 2017 in the reference area are not indicated with an asterisk. Adapted from: Hennekes et al., 2010.

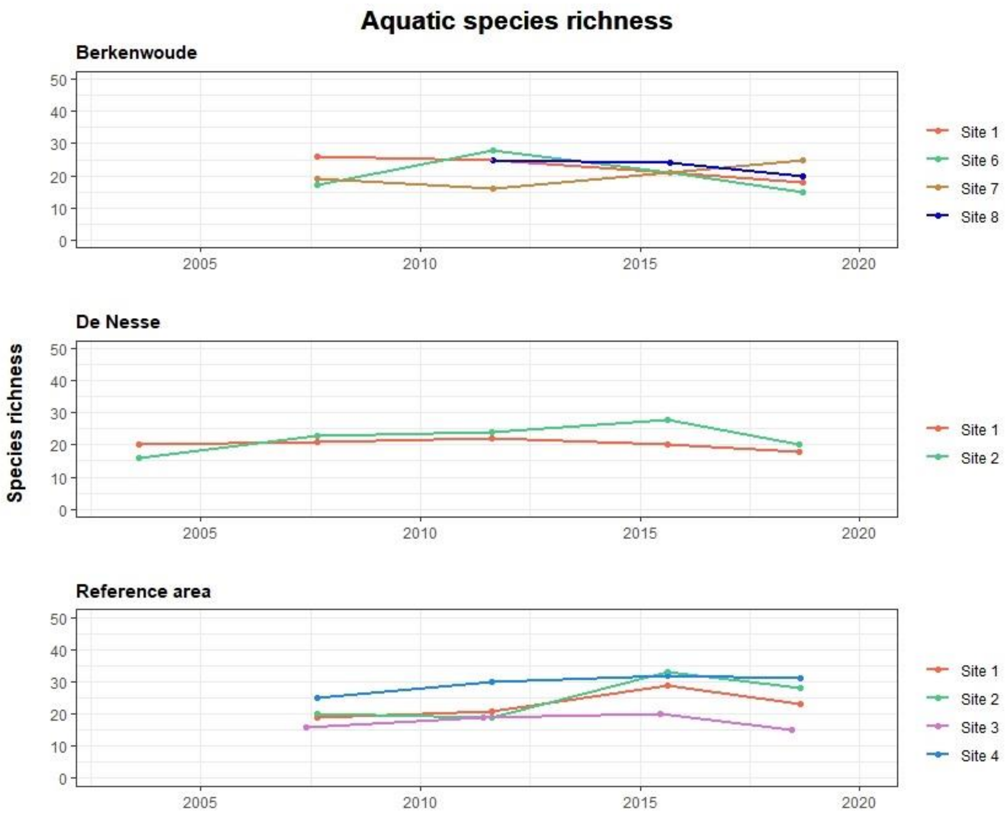

Figure 5 Species richness over time at the aquatic monitoring sites in Berkenwoude, De Nesse and the reference area. Site-specific colours correspond with the numbers of the monitoring sites as given in the methods section.

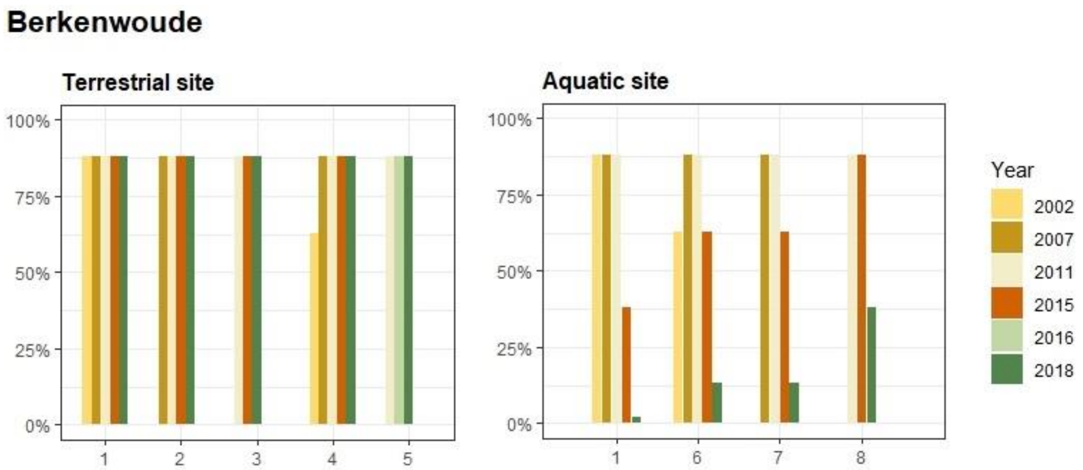

Figure 6 Total vegetation cover percentage at terrestrial and aquatic monitoring sites in Berkenwoude over time. The percentages are retrieved from the Braun-Blanquet method. A percentage of 88% corresponds with 5 on the Braun-Blanquet scale, indicating that 88% is the maximum cover percentage that can be achieved.

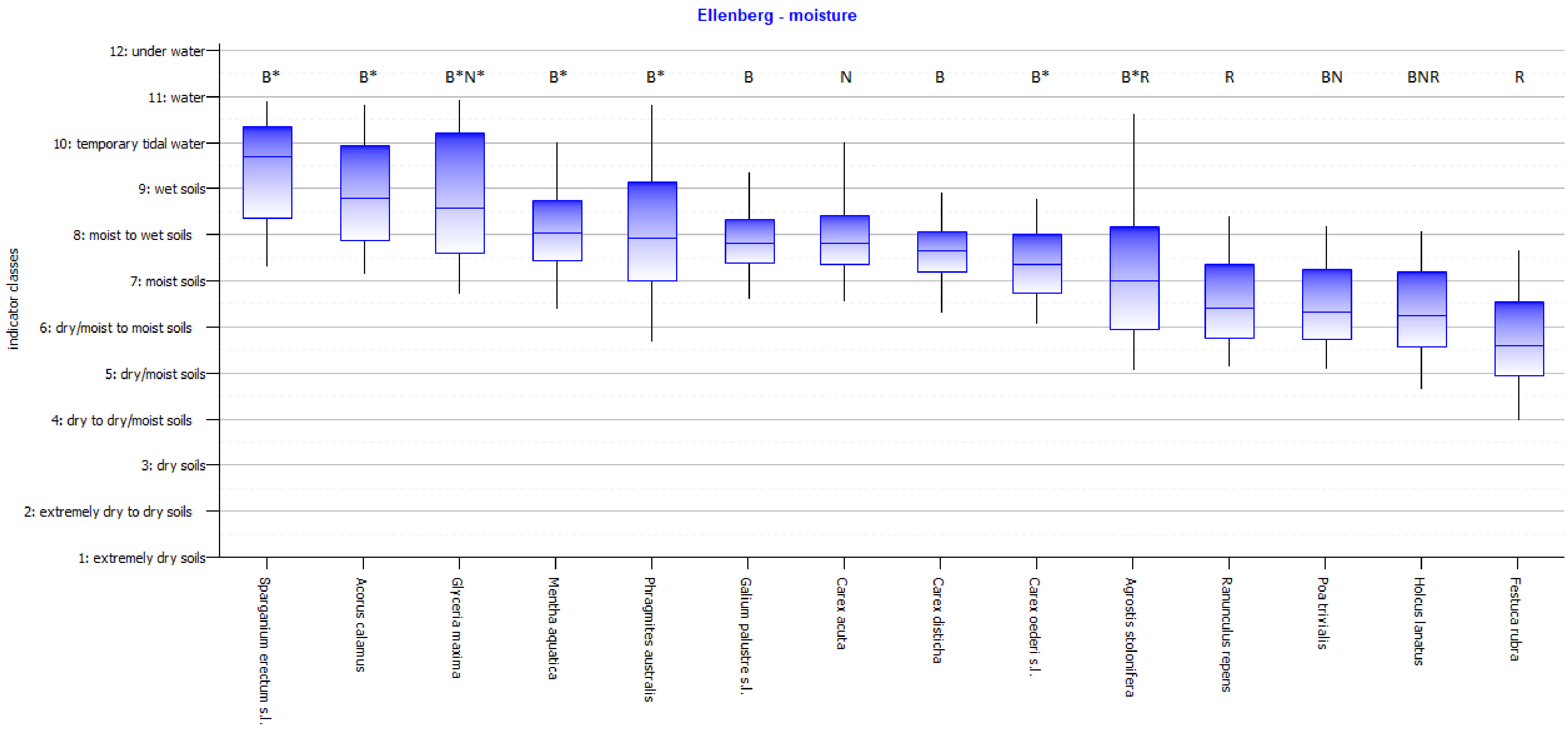

Figure 7 Ellenberg moisture classes of the species that were dominant at the terrestrial monitoring sites in the three areas – Berkenwoude (B), De Nesse (N), reference area (R). Species that were dominant after the water level increase in June 2017 in Berkenwoude and De Nesse are indicated with an asterisk. As water levels were not increased in the reference area, species that were dominant after June 2017 in the reference area are not indicated with an asterisk. Adapted from: Hennekes et al., 2010.

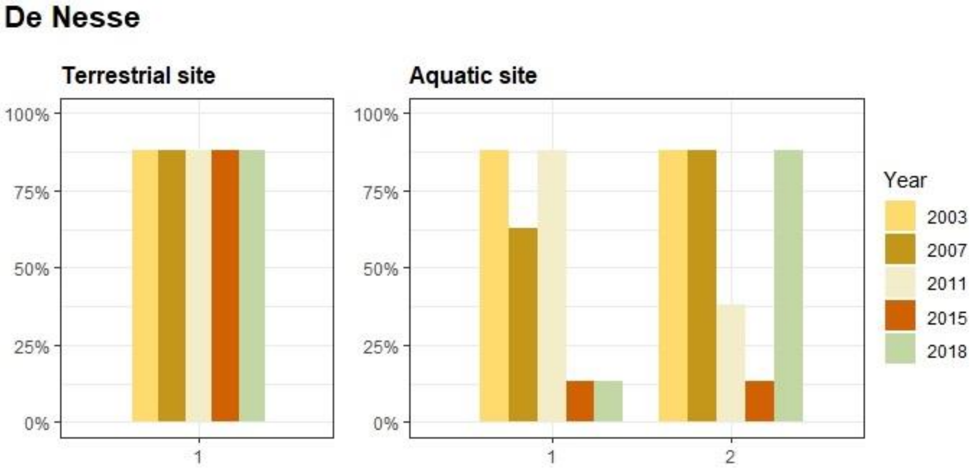

Figure 8 Total vegetation cover percentage at terrestrial and aquatic monitoring sites in De Nesse over time. The percentages are retrieved from the Braun-Blanquet method. A percentage of 88% corresponds with 5 on the Braun-Blanquet scale, indicating that 88% is the maximum cover percentage that can be achieved.

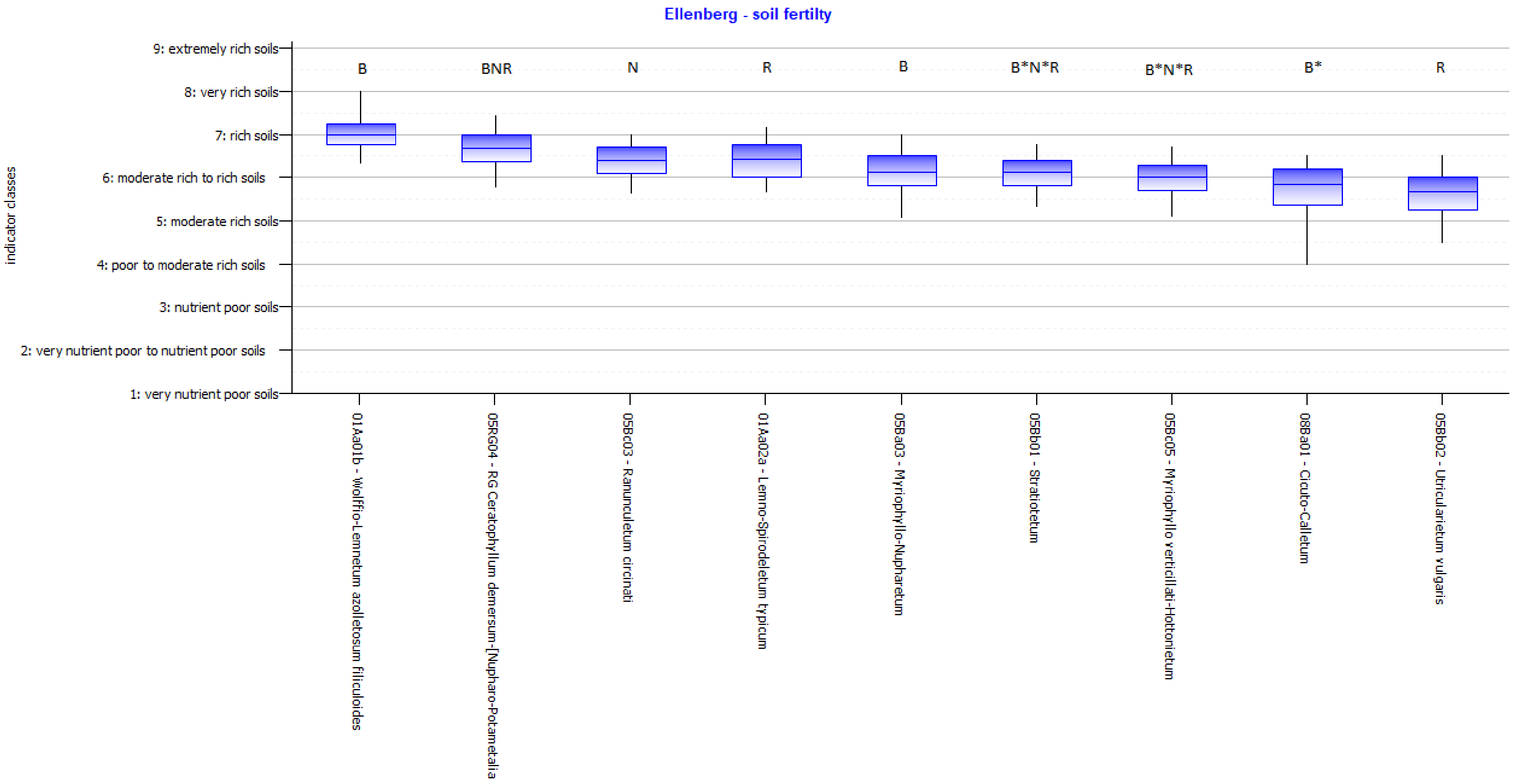

Figure 9 Ellenberg fertility classes of the vegetation types that were present at the aquatic monitoring sites in the three areas – Berkenwoude (B), De Nesse (N), reference area (R). Vegetation types that were present after the water level increase in June 2017 in Berkenwoude and De Nesse are indicated with an asterisk. As water levels were not increased in the reference area, vegetation types that were observed after June 2017 in the reference area are not indicated with an asterisk. Adapted from: Hennekes et al., 2010.

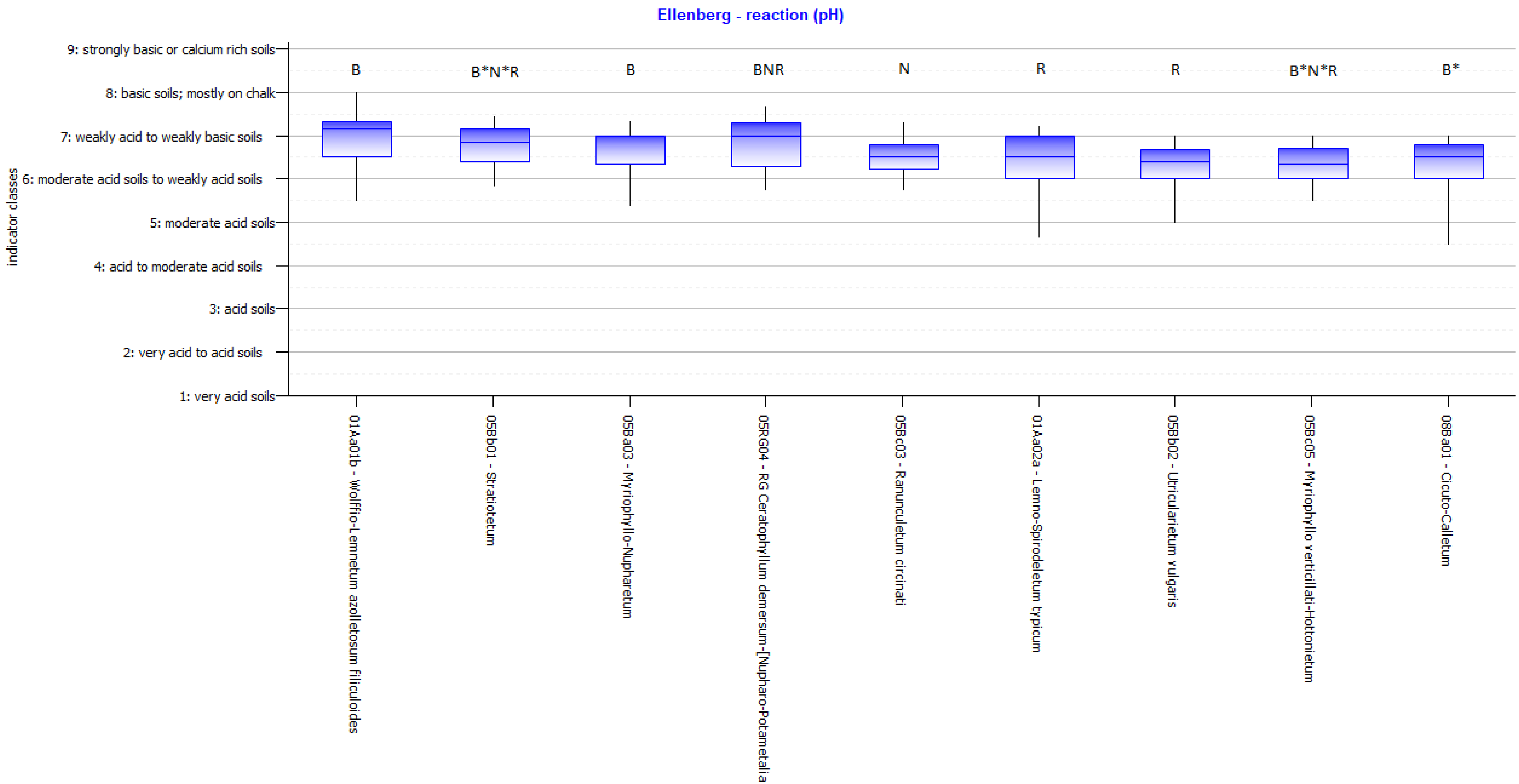

Figure 10 Ellenberg acidity classes of the vegetation types that were present at the aquatic monitoring sites in the three areas – Berkenwoude (B), De Nesse (N), reference area (R). Vegetation types that were present after the water level increase in June 2017 in Berkenwoude and De Nesse are indicated with an asterisk. As water levels were not increased in the reference area, vegetation types that were observed after June 2017 in the reference area are not indicated with an asterisk. Adapted from: Hennekes et al., 2010.

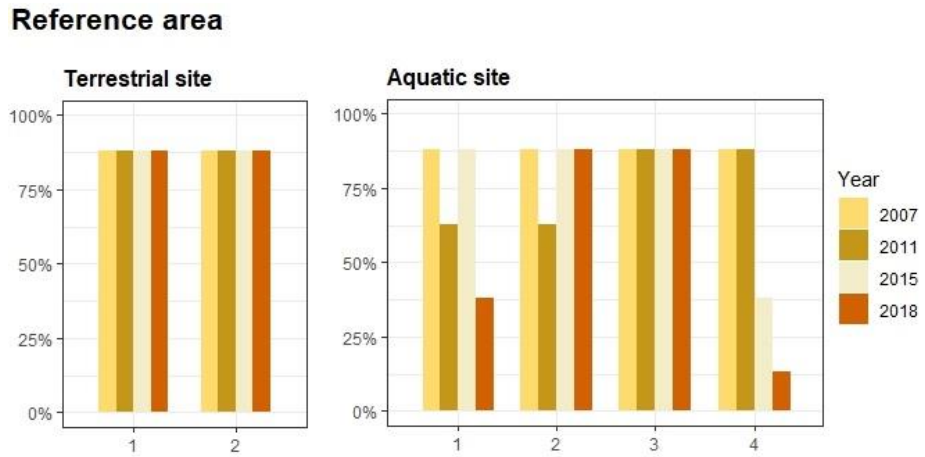

Figure 11 Total vegetation cover percentage at terrestrial and aquatic monitoring sites in the reference area over time. The percentages are retrieved from the Braun-Blanquet method. A percentage of 88% corresponds with 5 on the Braun-Blanquet scale, indicating that 88% is the maximum cover percentage that can be achieved.