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Monitoring the effects of ammonia air pollution with epiphytic lichens in the Netherlands, 1989–2022, a review

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Abstract. This paper deals with the results of long-term lichen monitoring in the Netherlands. We review the results related to ammonia (NH_3) air pollution over the period 1989–2022. NIW (nitrophyte abundance) and AIW (acidophyte abundance) were used as parameters to unite lichens with positive and negative responses to ammonia, respectively. Spatial patterns and temporal changes at large numbers of sampling sites with Oaks (*Quercus robur*) are compared and discussed. A steady decrease of acidophyte abundance (AIW) since 1989 is apparent, and continues up to now. The way in which AIW behaves strongly suggests that de-acidification of tree bark is the main cause of decline. This process is likely to be cumulative, i.e., proceeds as long as ammonia pollution continues. Recovery of bark acidity and acidophytic epiphytes is unlikely under current conditions. Loss of acidophytes is considered to be an important warning signal for habitat degradation. Changes in nitrophyte abundance (NIW) are not just the opposite of the AIW. Until 1998, there was a marked increase in nitrophyte abundance. After reaching its maximum in 1999, the NIW decreased again by 25–30%, probably as a result of measures to control the ammonia emissions. Nitrophytes had initially (<1999) increased fast due to the rapid de-acidification of tree bark. Many sites, at least in areas with high cattle densities, were fully de-acidified over the last 25 years. Measurements show that de-acidification eventually stops at a pH of about 6 or 6.5. Nitrophytes can be dominant at such sites, but can also disappear again, leaving a community with mainly crustose species typical for neutral bark. This is probably caused by a decrease in available nitrogen. Recent (since 2000) temporal changes of nitrophyte abundance (NIW) are likely reflecting changes in the load of ammonia.

Key words: acidophytes, ammonia, ecology, nitrogen, nitrophytes, temporal changes

Introduction

Lichen mapping and monitoring has been a common practice in the Netherlands for a very long time. Well known is the early work of Jan Barkman (1958) on cryptogamic epiphytes. He was the first to make a phytosociological survey across Europe and the Netherlands. De Wit (1976) presented maps of our country at 5 × 5 km^2 grid cells to show the impacts of sulphur dioxide (SO_2) air pollution based on wayside epiphytes. The recovery of epiphytes during the following decades was documented by Van Dobben (1993). He was the first to conduct a long-term monitoring program at fixed sites to show changes in species composition and correlations with abiotic factors. Monitoring of epiphytes in our coastal dune areas has been done by Aptroot et al. (2025) since 1990, with a new round to be carried out in 2026.

Van der Knaap (1980) was the first to notice an increase in nitrophytic lichens on trees near farms when surveying sites that were previously investigated by De Wit in 1973. The observations of Van der Knaap are the oldest written records that can be attributed to the impacts of gaseous ammonia (NH_3) on lichens. This was many years before ammonia air pollution was recognized by the Dutch authorities as a threat, and also long before it became feasible to measure ammonia air pollution with chemical methods.

At about the same time, the first author noticed a die-off of *Hypogymnia physodes* in the woods around his hometown, especially on *Larix* twigs (van Herk 2022). The cause was at that time not clear; however, it was obviously not related to the known changes due to sulphur dioxide. At that time, many farmers had switched the type of fertilization from a heavy straw mixed dung to a liquid urine mixed manure. This change of practice is important because the production of straw mixed dung involves access to oxygen (it is an aerobic process), while

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anaerobic conditions prevail when liquid manure is used, resulting in much higher reduced nitrogen (viz. ammonia) emissions.

This paper deals with results of long-term epiphyte monitoring since the 1990's. Three provincial authorities started in 1989/90 with a mapping and monitoring program focused specifically on the effects of ammonia. Within the six following years, four more provinces joined, and many 1,000s of sites were investigated, mainly in areas with nutrient-poor, sandy soils. A site typically consists of 10 selected trees, usually Common oaks (*Quercus robur*), often along a road. Tree trunks (0–2 m) are sampled individually and all lichen species are recorded on a six-point scale.

Two integrated parameters were developed (NIW and AIW), using lichens with a presumed positive response (nitrophytic species) and lichens with a presumed negative response (acidophytic species), respectively. Both parameters are calculated as the cumulative occurrence of about twenty selected species. NIW (and AIW) are defined as the mean number of nitrophytic (or acidophytic) species found per tree. Species covering more than 1 dm^2 per tree count as double. The species involved in calculating the NIW and AIW are summarized in Van Herk (1999).

Spatial patterns of NIW ('Nitrofiele Indicatie Waarde') were published. A striking resemblance between NIW and calculated ammonia air concentrations (based on cattle density and distance from sources) was evident. NIW and AIW ('Acidofiele Indicatie Waarde') were also related to in situ measurements of ammonia air concentration (using diffusion tubes) with comparable results ($r^2=0.59$). When NIW and NH_3 were aggregated to averages per 5 km grid cell, about 90% of the variance was explained (van Herk 2001).

Surprisingly, there appeared to be no correlation between NIW and actual or past sulphur dioxide levels, at least when using *Quercus* trees. It could be expected that areas with (former or current) high sulphur dioxide levels may have fewer nitrophytes (lower NIW), but this appeared not to be the case (van Herk 1999, 2001).

Chemical analysis of bark samples indicates that bark pH is very important for both acidophytic and nitrophytic lichen species. High ammonia levels may lead to an increase in the bark pH by more than two units due to the basic properties of ammonia. For instance, *Quercus* bark pH may rise from 3.8 to 6.2 as a result of this. This induces a shift from acidophytic to nitrophytic lichen assemblages. Ammonia adds to acidification when it is converted to nitrate (NO_3-) by nitrifying bacteria. However, such soil bacteria do not occur in tree bark.

In this context, it is important to note that naturally acid bark is, of course, not the same as acidified bark. However, tree species with naturally acidic bark may have a similar bark pH in both SO_2- polluted conditions (with a dominance of *Straminella conizaeoides*, syn. *Lecanora conizaeoides*) and unpolluted conditions (with luxurious *Usnea* communities). This was at least found for *Quercus*: in both mentioned lichen communities, where the bark pH was as low as 3.8 (van Herk 2001). The commonly used term 'acidified' can be misleading in this context because it implies that a lowering of bark pH has taken place due to acid pollutants, and this is supposed to be the prime cause of impoverishment of the lichen assemblages. Differences between these two acidophilous lichen assemblages, however, are far better explained by bark sulphate content. This is perhaps also the explanation of why nitrophyte abundances on *Quercus* bark were found to be similar in areas with high and low SO_2- pollution (van Herk 1999), i.e., nitrophytes appeared to be not less abundant at higher SO_2 levels. In both low and high SO_2- polluted areas the initial pH (i.e. with zero NH_3) is comparable, and perhaps, there is also a similar way in which the bark gets less acid due to the NH_3 deposition. For tree species with a naturally neutral bark, this is obviously different because the initial pH is not comparable.

Materials and methods

Nine out of twelve provinces in the Netherlands currently have a monitoring program on instigation of the provincial authorities, all following the fieldwork protocol designed by Van Herk (1999). Investigated provinces and years of survey [between square brackets the approximate number of sites] are: Drenthe (1991, 1998, 2004, 2010, 2016 & 2022) [560], Friesland (1991, 1996, 2003 & 2019) [250], Gelderland (1990, 1994, 1997, 2002, 2022 & 2024) [960], Groningen (1991, 1996 & 2022) [200], Noord-Brabant (1993 & 2024) [980], Overijssel (1989, 1994, 1999, 2005, 2015 & 2020) [720], Utrecht (1989, 1995, 2001, 2018 & 2025) [980], Zeeland (1997, 2000, 2003, 2006, 2019 & 2025) [270] and Zuid-Holland (2025) [450].

Results of Gelderland are showcased here as an example of how the data can be used by authorities for their policies about clean air management and the planning of nature conservation. Maps of NIW and AIW in 2022, and changes of NIW and AIW between 2002 and 2022 are presented, showing spatial patterns obtained with interpolated indication values (van der Kolk et al. 2022).

Overall trends of NIW and AIW (1989–2022) are calculated to present temporal changes of nitrophyte and acidophyte abundances for the Netherlands as an average. The changes were calculated with data from Drenthe, Friesland, Gelderland, Overijssel, Utrecht, and Zeeland. These six provinces, situated across the country, have sufficient continuity in research. Year-over-year changes were obtained with interpolation. When calculating the averages, all six provinces carry equal weight.

Results

Gelderland is a province with a high cattle density, while there are at the same time large protected nature areas that belong to the Natura 2000 framework of the European Union. Countries are obliged to adequately protect the biodiversity in these areas. Nitrogen deposition is potentially a major threat affecting these biodiversity goals. Lichen monitoring provides very detailed information on the degree to which certain Natura 2000 sites are affected by ammonia. An example is the Veluwe, a large (884 km^2) Natura 2000 site in the western part of Gelderland on nutrient-poor, coarse sandy soils. This forested area has drift sands, wet and dry heathlands, Juniper scrub, fens and raised bogs, all very susceptible to nitrogen deposition. NIW and AIW maps (Figs 1, 2) show where important ammonia sources are located, how deep the effects penetrate this nature area, what trends have occurred in recent years (Figs 3, 4) and how better protection can be achieved.

Temporal changes of NIW and AIW for the Netherlands as averages are shown in Figs 5 and 6. A steady decrease of acidophyte abundance (AIW) since 1989 is apparent, and this continues today. This process initially took place mainly in areas with high cattle densities and small woods, but expanded later to large wooded areas as well. Many sensitive species, such as *Hypogymnia physodes*, *Hypocenomyce scalaris*, and *Parmeliopsis ambigua*, have decreased sharply over the last few decades, while species such as *Ochrolechia microstictoides*, *Tuckermannopsis chlorophylla*, and *Platismatia glauca* are now almost extinct as epiphytes. Even common species like *Lepraria incana* have shown a significant decline in abundance. Nearly all the investigated sites show a decrease in AIW; increases occurred only sporadically (see Fig. 4). Losses are usually greatest at sites that have high potential for loss.

Changes of NIW are clearly different, not just the opposite of the AIW. Until 1998, there was a sharp increase in nitrophyte abundance. The NIW reached its maximum in 1999. Between 2000 and 2015, the NIW decreased again by 25–30%. Over the period 2016–2022, there was again a slight increase. The year 1999 was in several ways a tipping point. Until that time, NIW and AIW were correlated negatively, spatially as well as temporally. From 2000 onwards, this relationship got gradually blurred, mainly because acidophytes did not recover, while nitrophytes decreased at the same time.

Not all nitrophytes are represented evenly. *Polycauliona polycarpa* and *Physcia tenella* are 'early' species, popping up at low pollution levels and under relatively acid circumstances; *Phaeophyscia orbicularis* and *Candelariella aurella* can be found at high pollution levels, the first often abundant, the second in small numbers. Temporal trends can therefore vary significantly from species to species. A recent strong decrease of *P. polycarpa*, encountered widely over the country, can be explained by a sharp decrease in low-polluted, non de-acidified bark. This combination of factors was common around 1990, but is now rare.

Discussion

Environmental problems surrounding animal manure have been recognized by the Dutch authorities for over thirty-five years. Various measures have gradually been implemented to control the problem. Since the mid-1990s, it has been mandatory to incorporate manure directly

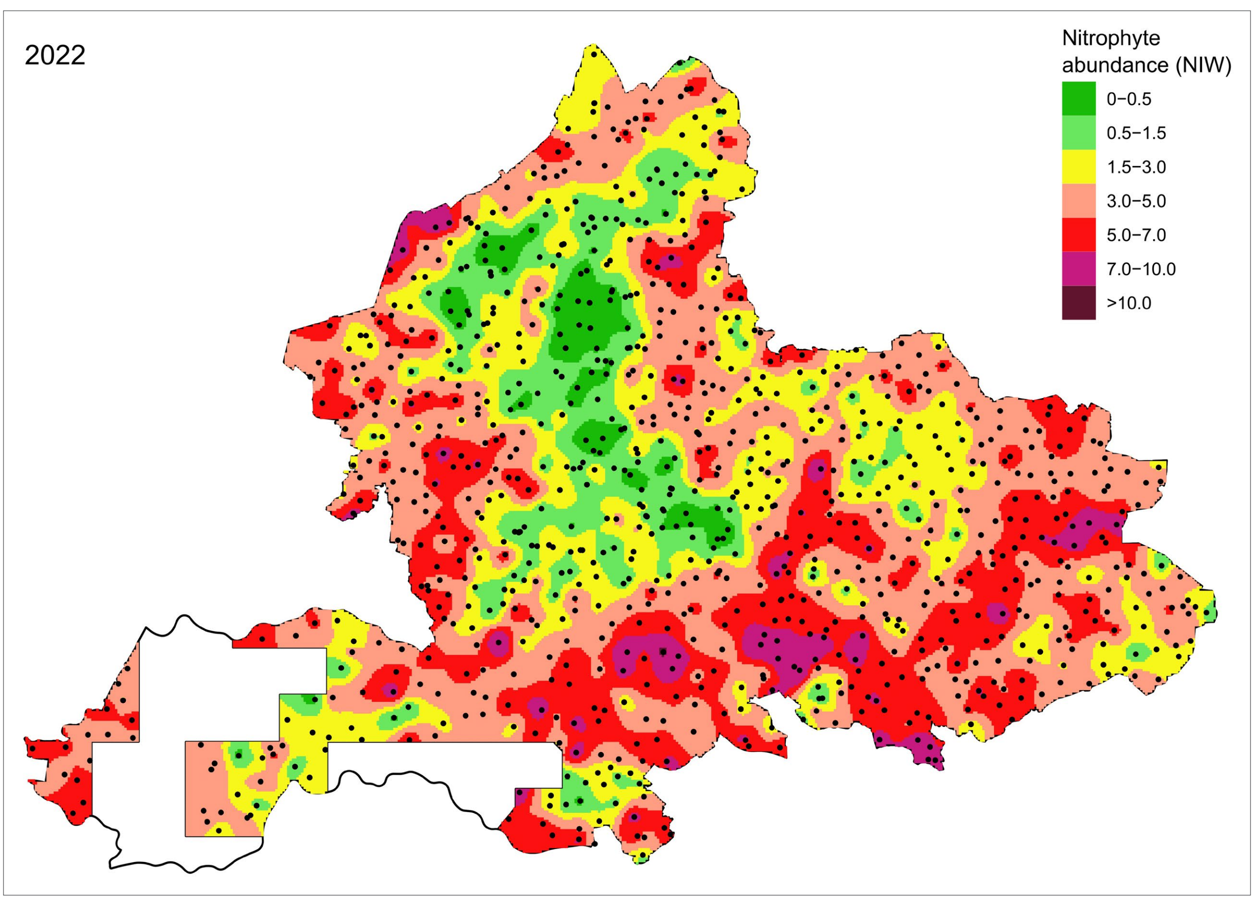

Figure 1. Patterns of abundances of nitrophytes (NIW) in Gelderland, 2022. Black dots are sample sites. High values (red and purple) correspond with a high nitrophyte abundance.

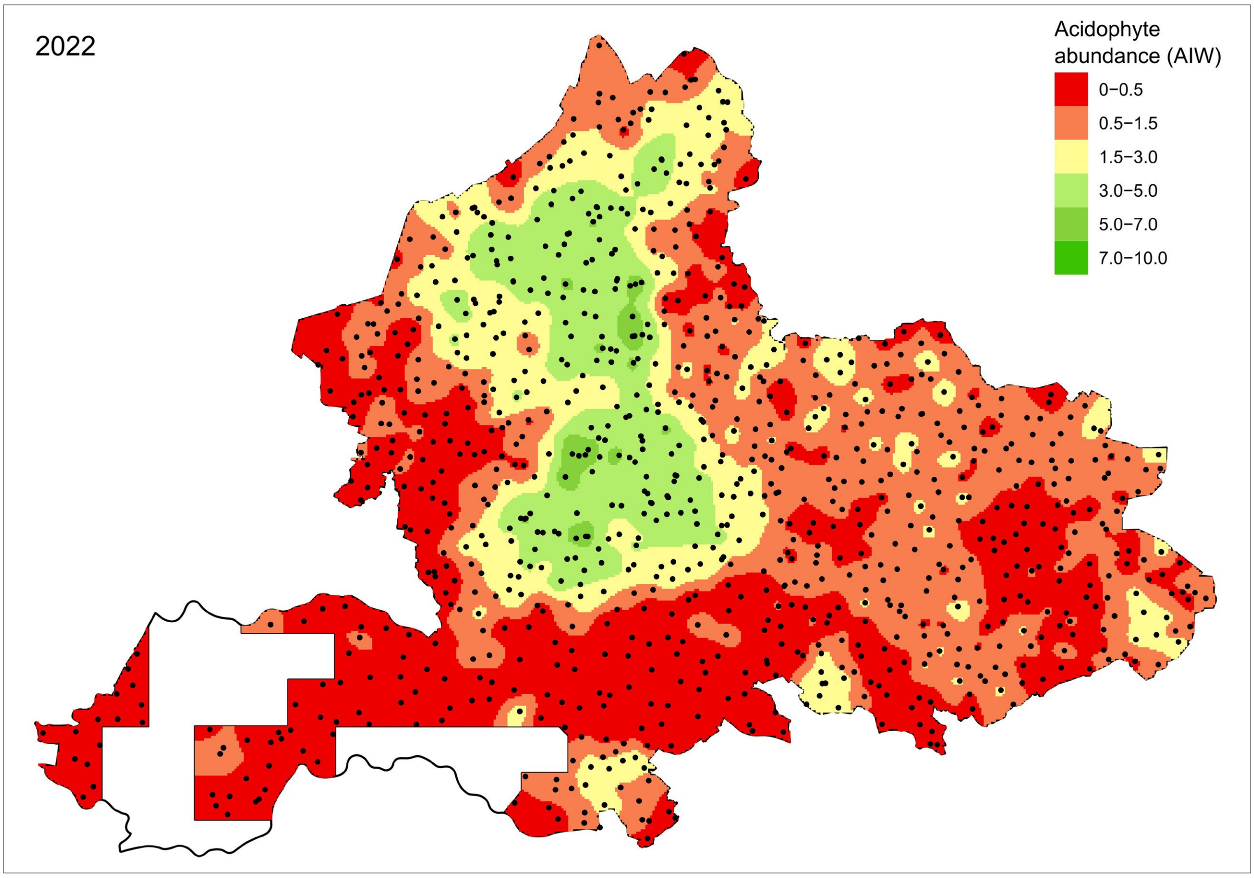

Figure 2. Patterns of interpolated abundances of acidophytes (AIW) in Gelderland, 2022. Black dots are sample sites. High values (green) correspond with many acidophytes.

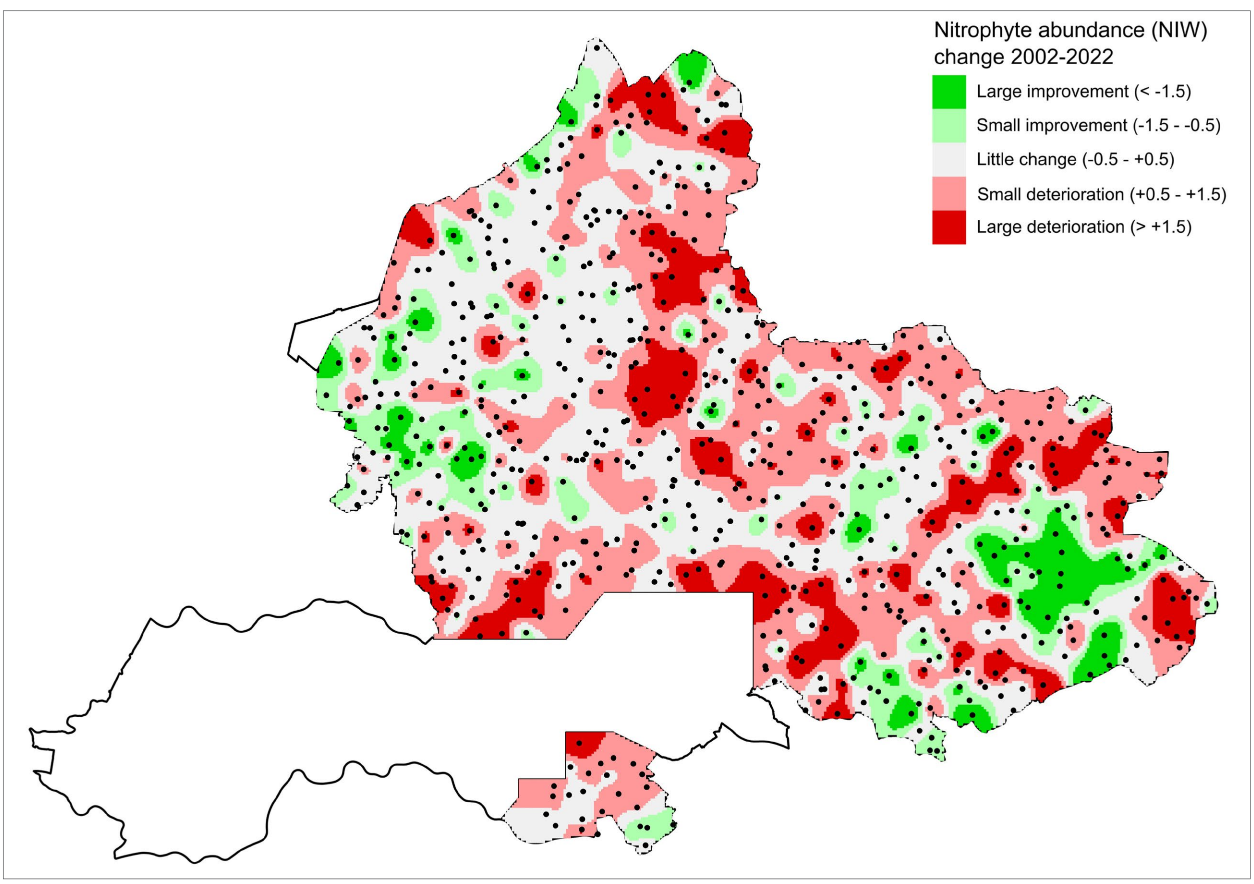

Figure 3. Change of spatial patterns of interpolated nitrophyte abundance (NIW) in Gelderland, 2002–2022. Red colors correspond with an increase of nitrophytes in that time period.

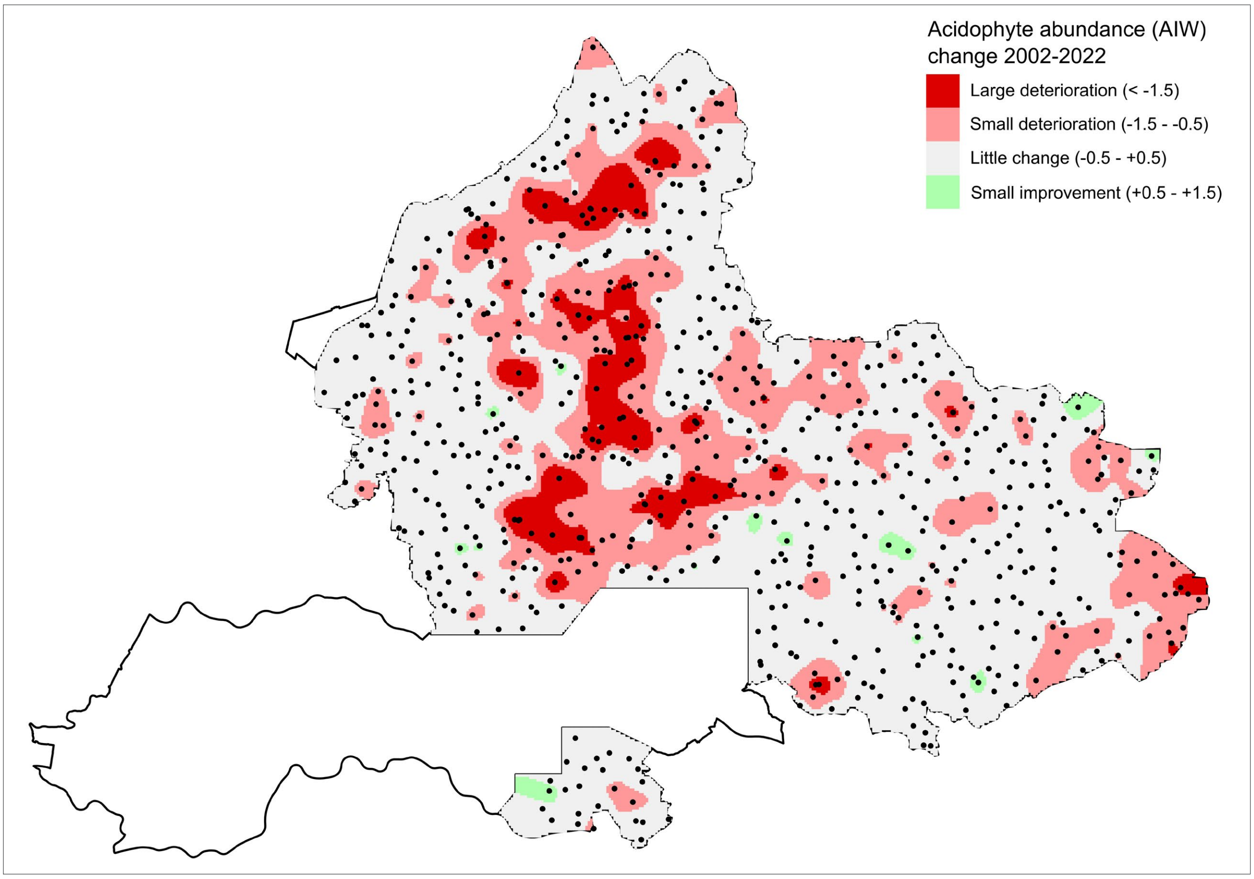

Figure 4. Change of spatial patterns of interpolated acidophyte abundance (AIW) in Gelderland, 2002–2022. Red colors correspond with loss of acidophytes.

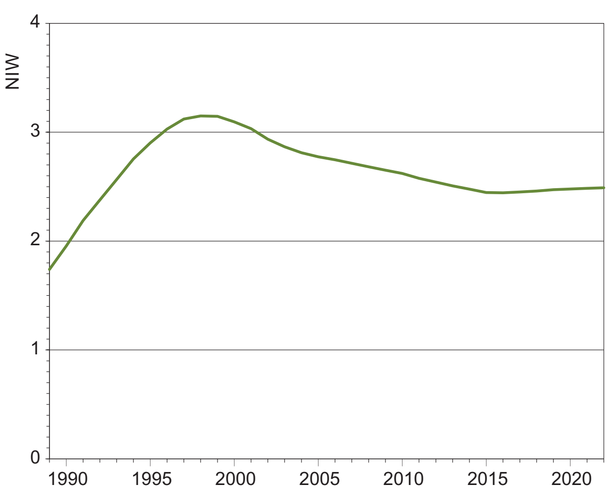

Figure 5. Temporal change of the abundance of nitrophytes (expressed as NIW) in the Netherlands, 1989–2022.

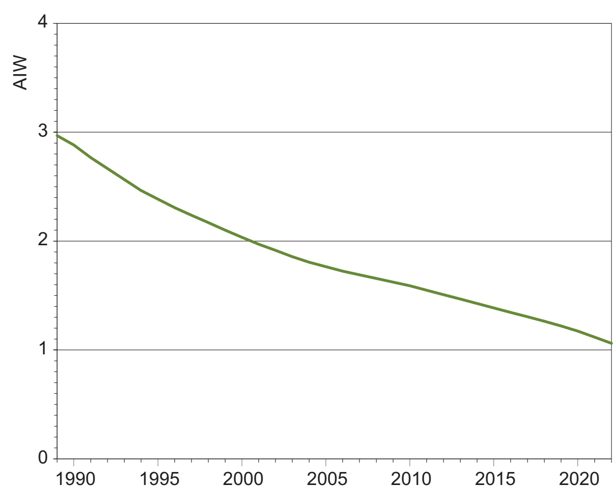

Figure 6. Temporal change of the abundance of acidophytes (expressed as AIW) in the Netherlands, 1989–2022.

into the soil (instead of spraying it on the surface) to reduce emissions. Manure silos must be covered, and more recently, new barn designs have been developed to limit ammonia emissions. Other measures include different, low-protein animal feed and experiments with the separation of urine and manure at the source. However, the environmental problems surrounding animal manure have not yet been resolved. The court has ruled now that most of our Natura 2000 areas are not sufficiently protected. It is up to the Dutch government at the moment to decide whether livestock population sizes should be reduced.

In light of this, it's important to understand exactly what the changes in NIW and AIW mean in terms of pollution trends. Patterns of NIW were shown to be coherent with NH_3 in space, but how is this with temporal trends? Several factors may be interfering, such as other pollutants and climate change. As a consequence of this, it can be very difficult to unravel the impacts of a single factor. NIW and AIW were designed to be balanced as much as possible, such that factors other than ammonia pollution would not impact them (van Herk 1999). For example, constituting species had a low sensitivity to SO_2. For different reasons, *Physconia grisea*, *Hyperphyscia adglutinata*, and

Candelaria concolor agg. were excluded from NIW. Climate change became important and led to an enormous increase in southern species (van Herk 2002), and it was therefore a good choice that these three warmth-loving species were not included in the NIW. If they had been included, the NIW would have become out of balance and a partial reflection of climate change. Although the effects of climate change are increasingly influencing species composition, there are up to now no indications for a substantial temporal imbalance in NIW or AIW.

Acidophyte abundance

Acidity was shown to be a primary factor in affecting the species composition by NH_3 (van Herk 2001). The way in which AIW behaves, however, strongly suggests that de-acidification is not only the main cause of decrease of acidophytes; it is likely to be a cumulative process as well (van Herk 2019), i.e., it proceeds further as long as pollution continues. Recovery of bark acidity and acidophytic lichens is unlikely given that current NH_3 levels are still high throughout the whole country. High concentrations are recorded in the air even in the middle of large nature reserves. If ammonia pollution disappears, the recovery would be quite unpredictable, but likely taking a long time. What we see is, in fact, the opposite of what occurred in the 1970s when all bark turned acidic due to high levels of SO_2, and remained acidic for a very long time, also when the SO_2 levels went down again. Acidity is therefore not a quantitative character that varies with the size of the load, but a condition that only ‘moves’ when a ‘force’ is exerted on it. This means that decreases in acidophyte abundance are hardly suitable to detect changes in levels of ammonia pollution.

At the same time, it is important to consider the overall biological impact of ammonia pollution. As cumulative effects are apparent on acidophytic lichens, this may be prevalent as well when considering in situ soil acidification, habitat degradation, loss of tree vitality, and shifts in terrestrial species composition. As a consequence, a fast loss of acidophytes is an important warning signal for the above-mentioned problems.

Nitrophyte abundance and de-acidification

Bark pH has been shown to be the limiting factor for nitrophyte abundance, at least at low pH. The ammonium (NH_4^+) content of bark by itself appeared not to be a decisive factor for nitrophytes under acidic conditions (van Herk 2001). The rapid increase of NIW between 1989 and 1998 deserves further explanation. It has been suggested that increases of nitrophytes at that time were just a result of lower SO_2 levels (e.g., van Dobben & ter Braak 1998). But the opposite might be true: what often has been regarded as a recovery due to dropping levels of SO_2, might be mainly the result of a de-acidifying process governed by NH_3. Tree bark doesn’t de-acidify by lower levels of SO_2 only. Earlier, it was discussed that spatial patterns of NIW didn’t correlate with SO_2, and pH of oak bark is similar in high and low SO_2- polluted areas. The rapid increase of NIW during that period is therefore regarded as a cumulative effect of high NH_3 levels (the switch from straw mixed dung to liquid manure was fully completed at that time), leading to a fast de-acidification, comparable to (but the reverse of) the behavior of the AIW. Similarly, increases in nitrophyte abundance, are at this stage, not necessarily indications of increasing ammonia pollution.

De-acidification is not only beneficial to nitrophytes. Other species may take advantage of this as well, such as many parmelioids and several *Ramalina* spp., at least if SO_2 air concentrations are low enough to prevent toxic effects. In general, ammonia pollution may lead to an overestimation of effects that are often attributed to SO_2 reductions.

However, in many sites, at least in areas with high cattle densities, tree bark has been de-acidified over the last 25 years. Oak bark has often lost its acidophytic species completely (even species that are relatively insensitive, such as *Lepraria incana*). Measurements show that de-acidification of oak bark eventually stops at a pH of about 6.0–6.5 (van Herk 2001; Boers 2023). Distant forest sites may still be in a ‘cumulative phase’ while at the same time many agricultural sites are fully de-acidified. Nitrophyte communities are sometimes well developed on de-acidified trees, but this is not necessarily the case. Monitoring has shown that nitrophytes can disappear on a large scale from de-acidified sites, leaving a community with species such as *Lecidella elaeochroma* and *Lecanora chlarotera*, i.e., a community of a-nitrophytic crusts typical for neutral bark. Few small thalli of *Xanthoria parietina* and *Physcia adscendens* are often the only remaining nitrophytes. These changes happened at a large scale between 2000 and 2015, resulting in a 25–30% decrease in NIW. A decrease of the NIW was, however, often also observed at sites which were only partly de-acidified. A die-off of Parmelioid species was first observed by Sparrius (2007) at monitoring sites where NIW had decreased. Species like *Parmelia sulcata*, *Melanelia subaurifera*, and *Ramalina farinacea* often disappeared where NIW suddenly became lower, especially at exposed sites. It was earlier discussed that similar species took advantage of increasing NH_3 levels.

The examples described above provide evidence that nitrophytes not only respond to an increase in bark pH. A major decline of the NIW has occurred between 2000 and 2015. Several measures to control the ammonia emissions (see earlier) reached their maximum effect during that period. The decline of NIW is likely an effect of the reduction of NH_3 emissions; however, this is difficult to prove. Decreased nitrophyte abundances have probably much to do with a decreased availability of nitrogen substances (Boers 2023), at least when acidity is not a limiting factor anymore. In their study in London, the availability of nitrogen was already suggested by Larsen et al. (2007) as a possible cause limiting the nitrophytes. Gadsdon et al. (2010), also studying the London area, found high NO_x concentrations to be an important predictor for high nitrophyte cover; however, no correlation was found when applying the NIW. Boers (2023), when repeating earlier research on bark properties by Van Herk (2001), found a strong correlation between bark N and NIW; however, no correlation was found with reduced nitrogen (NH_4^+) content. The strongest correlation was found by Boers with PM 2.5 (data obtained from the Dutch Air Quality Network), indicating nitrate (NO_3-) to be important. More research is needed to investigate the exact impacts of several nitrogen compounds, both reduced and oxidized . So far, it is not well understood which nitrogen substances are decisive, how they interact, and how they exactly impact the metabolism of the lichens.

In 2015, several rules were released for farmers in the Netherlands. Milk quotas were lifted, allowing them to keep more cows. In preparation for this, quite a few dairy farmers expanded their livestock between 2012 and 2015. Both new rules and the completion of the effect of the aforementioned measures might be an explanation for the (slight) increase or stable trend of the NIW between 2016 and 2022 instead of a further decline.

European perspective

Strong effects of ammonia pollution are observed over the whole of the Netherlands and eventually far beyond. Calculations have shown that approximately 25% of emitted NH_3 reaches a downwind distance of 100–1,000 km as ammonium (NH_4^+), and is deposited mainly in wet precipitation (Asman & van Jaarsveld 1990). In a study with ICP-IM monitoring stations across Middle and Northern Europe, it was shown that acidophilous species like *Bryoria fuscescens*, *B. capillaris*, and *Imshaugia aleurites* show negative correlations with ammonium content in precipitation (van Herk et al. 2003), also in regression together with other pollutants and temperature. Effects are discernible deep into the boreal coniferous zone of Scandinavia.

Future

Increasing evidence suggests that de-acidification appears to be a cumulative process, being persistent also when ammonia levels drop. A possible refugium for rare acidophilous species might be found in rough wooden poles and fences, including hardwood. For reasons that are not fully understood, this substrate sometimes remains remarkably well developed while trees in the surroundings have lost many of their sensitive species. Perhaps such wooden objects are buffered better against a rise in pH.

It is difficult to assess what is needed to protect the lichens of naturally acidic bark on a long-term scale. Now, it is likely that de-acidification, at best, can be slowed down. To reduce ammonia emissions, a return to the kind of farming of the 1960's with a heavy straw mixed dung would be preferable, but probably not feasible. Separation of dung and urine directly at the source is another option. Circular agriculture without net input of fertilizers and imported animal fodder, such as soy, is probably the best possible option for a sustainable future, but requires a completely different kind of livestock farming.

Conclusions

Huge temporal changes of nitrophytic and acidophytic lichen abundances in the Netherlands have occurred over the last 35 years. Many changes reflect changes in the load of ammonia pollution. Due to basic properties of ammonia, de-acidification of formerly acidic *Quercus* bark has occurred on a large scale. This process is probably cumulative, i.e., it proceeds further as long as ammonia pollution continues. Due to this, a steady decrease in acidophyte abundance since 1989 is apparent, and this will continue unless efforts to control emissions are implemented.

Nitrophyte abundance was initially (< 1999) also governed by pH as a limiting factor (at least on *Quercus*), however, many sites in areas with high cattle densities have been de-acidified over the last 25 years. Measurements show that de-acidification of *Quercus* bark eventually stops at a pH of 6 or 6.5. Monitoring shows that nitrophytes may disappear again, often leaving a community of mainly crustose species typical for neutral bark. Recent temporal nitrophyte decreases (>2000) are likely to reflect a decrease in ammonia. More research is needed to investigate the exact impacts of nitrogen. So far, it is not clear which nitrogen substances are decisive and how they interact.

Acknowledgements

We thank the provincial authorities of Drenthe, Friesland, Gelderland, Groningen, Noord-Brabant, Overijssel, Utrecht, Zeeland, and Zuid-Holland for making the lichen monitoring (1989–2026) possible.

It is no secret that André Aptroot is a prolific author in lichen taxonomy. Few people know, however, that he has contributed greatly to lichen ecology as well. His (co-authored) paper on 'global warming' in *The Lichenologist* in 2002 was the first worldwide in which temporal changes of a complete species assemblage could be attributed significantly to climate change. This paper addresses another field of his interest.

Author Contributions

Conceptualization, K.H.; fieldwork, K.H., H.K. and L.S.; formal analysis, K.H. and H.K.; writing: original draft preparation, K.H.; figures, H.K. and K.H.; writing, review and editing. K.H., H.K. and L.S.; funding acquisition, K.H. and L.S. All authors have read and agreed to the published version of the manuscript.

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