



## **Fire as a Restoration Tool in the Netherlands – First Results from Dutch Dune Areas Indicate Potential Pitfalls and Possibilities**

### **1. Introduction**

During the last three decades, the use of fire as a nature management tool in the Netherlands gradually lost ground to other management practices such as sod cutting, mowing and grazing. The increased nitrogen and acid deposition were the main reasons for this shift in management practice as these forced nature managers to implement more drastic measures to counteract the effects of eutrophication and acidification (Roelofs et al., 1993). Simultaneously, a gradual shift in both public and governmental opinion concerning the safety issues of the use of fire in nature reserves led to the abandonment of the use of fire as a nature management tool in the majority of nature reserves in the Netherlands. Nowadays, burning management in the Netherlands is only practiced in a small number of heathland areas in the south and central parts of The Netherlands. In the south, burning was never totally abandoned, although a transition from large scaled, intensive burning management towards local, extensive burning management occurred over time. More intensive use of burning management is still practiced in the central parts, on military training areas located in the Veluwe region. Due to the explosion risk of military projectiles present in the soil, the use of fire as a means to prevent succession towards forest is the only safe method that can be used (van der Zee, 2004).

Recently, burning management received renewed attention from both nature managers and ecologist as a viable alternative restoration management practice. It became clear that the success of those management practices that focused solely on the -removal of nutrients had its limitations. For instance, although sod cutting in grass-encroached heathlands did result in the successful re-establishment of *Calluna vulgaris*, other target-species showed little or no positive response. Furthermore, the decline of animal species characteristic of dry nutrient poor ecosystems was not stemmed by the restoration efforts taken during the last two decades (Stuijzand et al., 2004).

Recent experiments with liming of sod-cut heathlands showed that restoration of the soil buffering capacity was an important factor contributing to the success of restoration. A remarkable increase in the germination and settlement of highly endangered plant species (such as Mountain Arnica, *Arnica montana*) was observed (De Graaf et al., 1998). Positive effects of prescribed burning on both plant (e.g., *Arnica montana* [Hornman and Haveman, 2001]) and animal (e.g., the highly endangered Heath Bush-cricket, *Gampsocleis glabra* [van der Berg et al., 2000]) species composition in heathlands on the military firing ranges raised the question whether burning is a viable alternative restoration measure in nutrient-poor ecosystems. Following these experiences prescribed burning management was considered again as a potential nature management tool in the Netherlands.

In addition to heathlands on nutrient poor sandy soils, coastal dunes are highly sensitive to increased atmospheric deposition as well. Increased nitrogen deposition levels in Dutch dune areas, varying between 10 and 25 kg/ha/year, resulted in a significant increase in available nitrogen in these systems. As a consequence, open dune grasslands such as lichen rich *Corynephorus* grasslands and forb-rich *Festuca* grasslands (grey dunes) have been seriously degraded as a result of grass-encroachment by fast growing species such as Sand Sedge (*Carex arenaria*), Wood Small-reed (*Calamagrostis epigejos*) and Marram Grass (*Ammophila arenaria*) and/or shrub encroachment by Sea Buckthorn (*Hippophae rhamnoides*), Rum Cherry (*Prunus serotina*) and Elderberry (*Sambucus nigra*) (Kooijman et al., 1998). In addition, acidification results in the inhibition of the decomposition of organic material (Delaune et al., 1981). Combined with the increased litter production by the abovementioned tallgrass species, this results in the formation of thick humus and litter layers, which in turn hamper the germination of characteristic plant species, in particular forbs and annuals. The relevant habitat conditions (warm and dry habitats) for animal species are severely altered as well, resulting in a decline in species characteristic of dry dune grasslands, and a general increase in eurotopic, mesophylic species. In the case of shrub-encroachment, shading of the soil inhibits germination of characteristic plant species and severely alters the microclimatic conditions of the

habitat, which is thought to be one of the main causes for the decline of characteristic fauna species in shrub-encroached habitats (van Turnhout et al., 2003).

To date, restoration management of degraded dry dune grasslands has had only limited success. Due to the hilly terrain characteristics, widespread use of mechanical means of vegetation and/or top soil removal is both impractical and expensive. During the last three decades, grazing by large herbivores has been widely implemented as a management tool to counteract the process of grass- and shrub-encroachment and reduce the accumulation of litter and humus. Although this management proved to be quite effective in inhibiting further spread of grass-encroachment and/or shrub-encroachment, dry dune grasslands that were already in a degraded state before grazing was implemented show less pronounced signs of recovery (Kooijman and van der Meulen, 1996).

In order to improve the effectiveness of restoration management of dry dune grasslands, burning has gained renewed attention by nature managers and scientists. Possible benefits of using prescribed burning include:

- a) selective removal of accumulated nutrients such as nitrogen and sulphur by means of evaporation and/or oxidation of these elements in gaseous form (Neary et al., 1999);
- b) when executed in the winter period; a relatively low impact on ecosystem functioning, compared to more drastic measures such as sod cutting;
- c) a temporary increase in soil buffer capacity as a result of the deposition of cations from ash (Raison, 1979);
- d) an increase in the nutritional quality of the vegetation post-burn as a result of both quantitative (higher ratio of standing crop/standing dead; higher leaf/stem ratio, shifts in vegetation composition) as well as qualitative characteristics (an increase of macronutrient and/or protein content in living biomass) of the vegetation (Gimingham, 1972; Van de Vijver et al., 1999); and
- e) higher cost-efficiency of the measure compared to other means of biomass removal.

In order to assess the viability of burning as a restoration tool, The Dutch State Forestry Service performed some small-scale experiments with prescribed winter burning in grass-encroached dune grasslands on the Wadden Sea Islands. In the same years, a number of summer wild-fires occurred in dune areas in both the Wadden sea area as well as the lime richer dune area on the mainland. This provided the opportunity to compare the effects of the experimental small-scaled fires to relatively large scaled, high intensity fires.

The main research-question of this first investigation was if prescribed burning could be used as an effective tool to remove accumulated nitrogen by means of combustion of the litter and humus layers. The amount of litter and humus removed by burning, the rate of regrowth of the vegetation and accumulation of litter, as well as differences in plant species composition before and after burning were investigated. Secondly, the effect of burning management on animal species composition was assessed by investigating the carabid beetle assemblages before and after burning. Research on this topic focused primarily on the recovery rate of characteristic species and assessed whether this recovery was mainly driven by survival or (re)colonization of species.

## 2. Study sites and methods

Prescribed burning experiments were carried out on the Wadden sea islands of Ameland and Terschelling (PB1 and PB2 respectively, see Figure 1) in the winter of 2003 and 2004. Burnt plots were small; about 0.5 ha surface area per plot. Before burning took place, fire-breaks were created around the plots by means of mowing. During and several days prior to burning, temperatures were below zero. Prescribed burning took place in the afternoon, using a headfire at wind speeds between Bft 1 (Terschelling) and 4 (Ameland).

Wildfires occurred on Terschelling in 2004 on 15 May and in the lime rich mainland-dunes near Castricum on 9 August (WF1 and WF2 respectively, see Figure 1). Due to the extreme dry weather conditions in both cases and the large scale of both wildfires (55 and 20 ha respectively), the intensity of these wildfires was much higher than the prescribed fires.

The vegetation type of the prescribed fire sites was grass-encroached, dry dune grassland, dominated by *Ammophila arenaria* and *Carex arenaria*. Vegetation type of the wildfire-site on Terschelling was a mixture of grass-encroached, *Ammophila arenaria*-dominated vegetation, intact *Corynephorus*

*canescens* dominated dry dune grassland and dune heathland dominated by *Empetrum nigrum*. The Castricum wildfire site consisted of a mixed vegetation of grass-encroached *Calamagrostis epigejos* dominated dune grassland and shrub-encroached vegetation dominated by *Hippophae rhamnoides*.

At all experimental sites, the biomass of herb, moss and litter layers was assessed by means of collecting biomass samples of all three layers before and after prescribed burning. In the case of the wildfires, biomass samples of all three layers were collected at the same time in burnt and nearby unburned control-sites. On sites PB1 and WF1, samples were taken both directly after burning took place, as well as one year after burning in order to assess the speed of vegetation recovery. On site PB1, samples were taken only directly after burning took place. On site WF2 biomass samples were taken one year after burning took place. The carabid beetle assemblage was assessed by means of pitfall-trapping on the sites PB1; WF1 and WF2. Pitfall traps were placed in the burnt areas and in nearby unburned vegetation, which was used as a control.

### Statistical analysis

Statistical analysis of data was performed using SPSS 13.0 for Windows (SPSS 2004) Difference in mean dry weight of each separate layer between treatments was tested for significance using independent samples t-tests. When assumptions on normality of data and/or homogeneity of variance were violated, Mann-Whitney tests were used. Carabid beetle assemblage was compared between treatments using the classification program Twinspan (Hill, 1979). At the Castricum wildfire site, differences in proportions between total flight capable and flight incapable carabid beetles were tested for significance using Pearson's chi square test for independence.



**Figure 1.** Geographical location of the study sites. Prescribed burning-experiments were executed on Ameland (PB1) and Terschelling (PB2), wildfires occurred on Terschelling (WF1) and the dune area of Castricum (WF2). In all sites, effect of burning on living and dead biomass was assessed. Carabid beetle assemblage was investigated in PB1, WF1 and WF2.

### 3. Results

#### Biomass removal

In all study sites, burning resulted in significant losses in standing biomass (Table 1). Total biomass of moss and litter layers in the prescribed burning study sites was unaffected by burning treatment. In contrast, biomass of moss (*Empetrum nigrum*) and litter (*Ammophila arenaria*) layers was significantly reduced in the wildfire study site on Terschelling (WF1). Although no moss layer was present in the burnt treatment in the *A. arenaria* site, it did not differ significantly to the control site as a result of high variability in total biomass in the control vegetation. Biomass of litter did not differ between both treatments in the *E. nigrum* site. In the burnt site, litter consisted mainly of ash deposits and incompletely burnt remnants of the previous vegetation (visual assessment), suggesting that a substantial combustion of litter that was present before burning did indeed take place.

**Table 1.** Mean biomass (g/m<sup>2</sup>) and significance of difference between burnt and control vegetations, subdivided into standing crop, moss and litter layers, directly after burning.

| Area   | Vegetation type   | Layer    | Treatment | N       | Mean biomass | St. error of mean | Significance of difference |
|--------|---|----------|-----------|---------|--------------|-------------------|----------------------------|
| PB1    | <i>Calamagrostis epigejos/Carex arenaria/Ammophila arenaria</i> dominated | standing | burnt     | 16      | 39.97        | 5.59              | p<0.001                    |
|        |   |          | control   | 16      | 354.33       | 31.76             |                            |
|        |   | moss     | burnt     | 16      | 191.51       | 50.24             | N.S.                       |
|        |   |          | control   | 16      | 216.73       | 41.53             |                            |
|        |   | litter   | burnt     | 16      | 414.66       | 67.14             | N.S.                       |
|        |   |          | control   | 16      | 382.28       | 68.17             |                            |
| PB2    | <i>Ammophila arenaria</i> dominated                                       | standing | burnt     | 4       | 179.74       | 32.86             | p<0.01                     |
|        |   |          | control   | 4       | 421.60       | 47.09             |                            |
|        |   | moss     | burnt     | 4       | 13.08        | 13.08             | N.S.                       |
|        |   |          | control   | 4       | 59.96        | 21.54             |                            |
|        |   | litter   | burnt     | 4       | 264.28       | 53.73             | N.S.                       |
|        | control   |          | 4         | 685.92  | 209.68       |                   |                            |
|        | <i>Empetrum nigrum</i> dominated  | standing | burnt     | 4       | 127.24       | 58.60             | p<0.001                    |
|        |   |          | control   | 4       | 1727.76      | 194.64            |                            |
|        |   | moss     | burnt     | 4       | 0.00         | 0.00              | N.S.                       |
|        |   |          | control   | 4       | 26.4         | 15.31             |                            |
| litter |   | burnt    | 4         | 392.12  | 74.58        | N.S.              |                            |
|        | control   | 4        | 633.00    | 69.61   |              |                   |                            |
| WF1    | <i>Ammophila arenaria</i> dominated                                       | standing | burnt     | 4       | 82.76        | 10.97             | p<0.01                     |
|        |   |          | control   | 4       | 749.20       | 165.87            |                            |
|        |   | moss     | burnt     | 4       | 0.00         | 0.00              | N.S.                       |
|        |   |          | control   | 4       | 290.16       | 142.46            |                            |
|        |   | litter   | burnt     | 4       | 62.68        | 28.68             | p<0.01                     |
|        | control   |          | 4         | 228.28  | 24.64        |                   |                            |
|        | <i>Empetrum nigrum</i> dominated  | standing | burnt     | 8       | 210.72       | 47.55             | p<0.001                    |
|        |   |          | control   | 8       | 776.96       | 85.04             |                            |
|        |   | moss     | burnt     | 8       | 0.38         | 0.38              | p<0.001                    |
|        |   |          | control   | 8       | 287.10       | 85.49             |                            |
| litter |   | burnt    | 8         | 165.56  | 35.05        | N.S.              |                            |
|        | control   | 8        | 234.44    | 103.351 |              |                   |                            |

#### Recovery of vegetation

After six months; mean biomass of standing crop, moss and/or litter layer did not differ significantly between burnt and control treatments at the experimental burning site on Ameland (PB1; Table 2). The wildfire site on Terschelling (WF1; Table 2) still showed significant differences in total biomass after one and a half year. Biomass of all three layers in the *E. nigrum* vegetation was significantly lower compared to the control. In the *A. arenaria* dominated vegetation however, mean biomass only showed significant differences in the litter layer. This was largely a result of fast vegetative recovery of *A. arenaria* from (belowground) rhizomes and/or root systems that survived the fire. More importantly, this effect was also pronounced in the burnt *Empetrum nigrum* dominated sites. Fast vegetative

regrowth of *A. arenaria* resulted in a shift from *E. nigrum* dominance in the pre-burn state into *Ammophila arenaria* dominance post burn.

One year after the fire, at the wildfire site near Castricum (WF2), significant differences in mean biomass were only found in the moss layer. Reduction in litter biomass was near significant; biomass of standing crop showed a recovery towards pre-burn conditions within one year.

**Table 2.** Mean biomass (g/m<sup>2</sup>) and significance of difference between burnt and control vegetations, subdivided into herb, moss and litter layers, 6 months (PB1); 1.5 years (WF1) and 1 year (WF2) after burning.

| Area   | Vegetation type   | Layer    | Treatment | N      | Mean biomass | St. error of mean | Significance of difference |
|--------|---|----------|-----------|--------|--------------|-------------------|----------------------------|
| PB1    | <i>Calamagrostis epigejos/Carex arenaria/Ammophila arenaria</i> dominated | standing | Burnt     | 16     | 310.63       | 46.65             | N.S.                       |
|        |   |          | Control   | 8      | 417.36       | 55.51             |                            |
|        |   | moss     | Burnt     | 16     | 335.86       | 41.27             | N.S.                       |
|        |   |          | Control   | 8      | 319.70       | 87.19             |                            |
|        |   | litter   | Burnt     | 16     | 246.29       | 51.67             | N.S.                       |
|        |   |          | Control   | 8      | 328.98       | 105.15            |                            |
| WF1    | <i>Ammophila arenaria</i> dominated                                       | standing | Burnt     | 4      | 688.52       | 244.76            | N.S.                       |
|        |   |          | Control   | 4      | 639.52       | 118.03            |                            |
|        |   | moss     | Burnt     | 4      | 349.00       | 344.95            | N.S.                       |
|        |   |          | Control   | 4      | 340.16       | 195.79            |                            |
|        | litter  | Burnt    | 4         | 150.32 | 6.37         | p<0.05            |                            |
|        |   | Control  | 4         | 695.96 | 109.26       |                   |                            |
|        | <i>Empetrum nigrum</i> dominated  | standing | Burnt     | 8      | 157.48       | 31.68             | p<0.001                    |
|        |   |          | Control   | 8      | 882.50       | 132.15            |                            |
|        |   | moss     | Burnt     | 8      | 2.54         | 1.485             | p<0.001                    |
|        |   |          | Control   | 8      | 352.56       | 82.11             |                            |
| litter | Burnt   | 8        | 43.12     | 7.728  | p<0.001      |                   |                            |
|        | Control   | 8        | 180.26    | 25.56  |              |                   |                            |
| WF2    | <i>Calamagrostis epigejos/Hippophae rhamnoides</i> dominated              | standing | Burnt     | 10     | 174.93       | 22.56             | N.S.                       |
|        |   |          | Control   | 10     | 277.49       | 75.19             |                            |
|        |   | moss     | Burnt     | 10     | 0.00         | 0.00              | p<0.001                    |
|        |   |          | Control   | 10     | 414.50       | 76.85             |                            |
|        |   | litter   | Burnt     | 10     | 53.15        | 15.10             | N.S.                       |
|        |   |          | Control   | 10     | 123.98       | 42.10             |                            |

### Short term effects of burning on carabid species composition

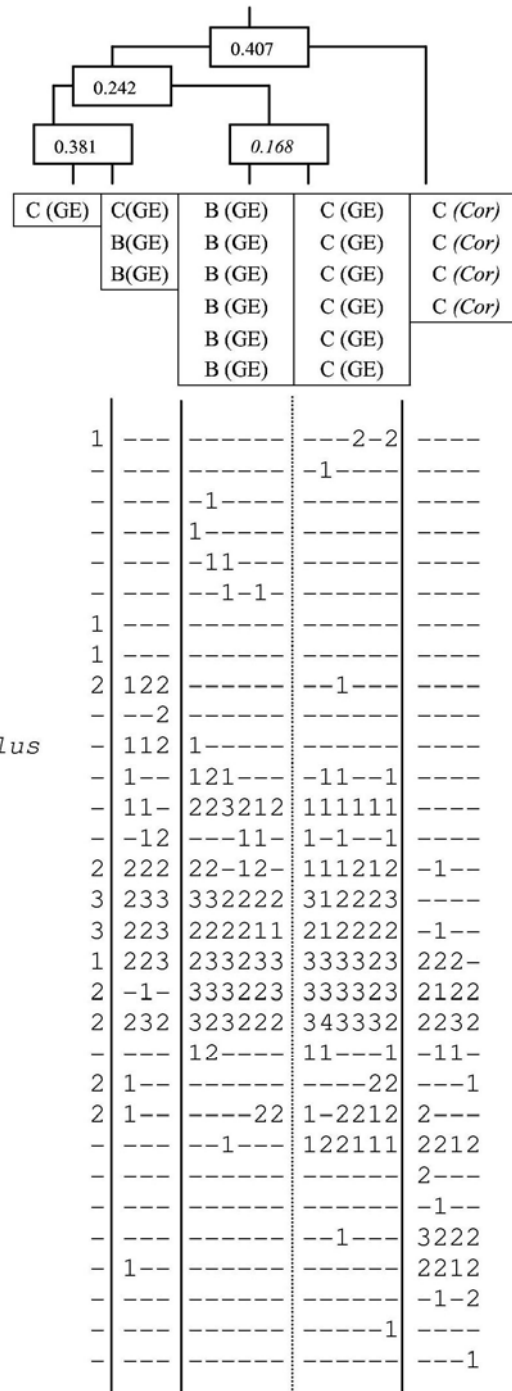
#### Experimental site: Ameland (PB1)

Two months after the execution of burning, the species assemblage of the carabid beetle community showed a weak affinity with the burnt and unburned control sites (Figure 2). The most pronounced difference however, existed between unburned, *Corynephorus canescens* dominated vegetation and all grass-encroached vegetations, independent on whether burning did or did not take place in those vegetation types (first division, eigenvalue of 0.407). Other differences in species composition between sampling sites were indifferent on whether burning took place (second and third division, eigenvalues 0.242 and 0.381 resp.) or showed a weak, but consistent difference between burnt and control sites (fourth division, eigenvalue 0.168).

Six months after burning (Figure 3), significant difference in species assemblage was only found between unburned *Corynephorus canescens* dominated vegetation (including one control grass-encroached site) and all other sites, independent on whether that site had been subject to burning management or not (first division, eigenvalue 0.250).

#### Wildfire site: Terschelling (WF1)

Total number of species captured at the wildfire site was low compared to the other investigations. The species assemblage on the wildfire site at Terschelling showed some reaction to burning (Figure 4), however, the vegetation present prior to burning also had a large impact on the species composition after burning. Burnt *Corynephorus canescens* vegetation differed most in species composition with the other sites (first division, eigenvalue 0.284). Further divisions were made between burnt and unburned



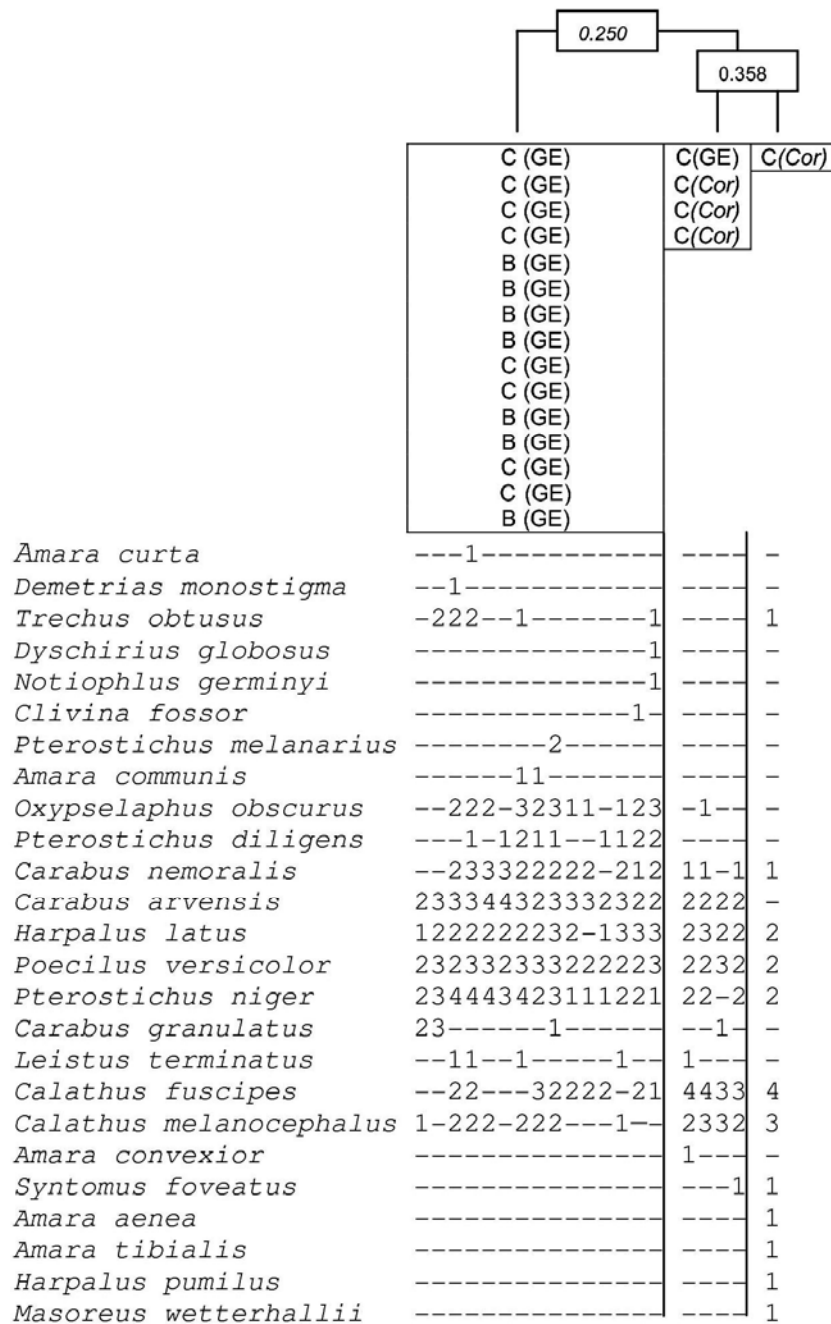
**Fig 2:** Twinspan dendrogram of Carabid beetle assemblage of the experimental burnt site at Ameland (PB1), 2-3 months after burning took place. B= burnt site; C= control site. Abbreviations between brackets: GE= Grass encroached vegetation; Cor=Corynephorus canescens dominated vegetation. Numbers shown at each division are eigenvalues (between 0.0-1.0) associated with that division. Scores associated with carabid species are abundance classes from lowest (1) to highest (9).

vegetations and former *E. nigrum* dominated vegetation and *Ammophila arenaria* dominated vegetation (second and third division; eigenvalues 0.307; 0.211 resp.).

Wildfire site: Castricum (WF2)

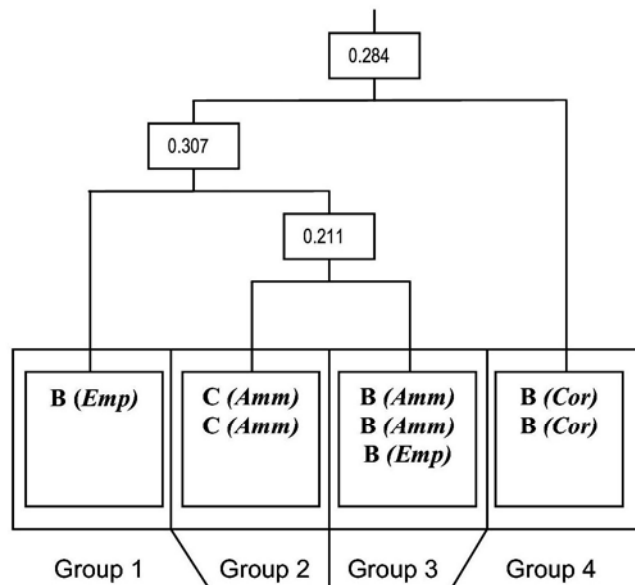
Total number of species trapped was much higher compared to the other investigation areas. The species assemblage was clearly affected by burning (Figure 5); most of the unburned sites are separated from the burnt sites (first and fifth division, eigenvalues 0.301; 0.300 resp.). Total number of species trapped in the burned sites was higher than in the unburned control sites. Furthermore, a clear

distinction could be made between species with flight capability and species which do not have flight capability. Flight capable species were mostly responsible for the divisions. For instance: group



**Fig 3:** Twinspan dendrogram of Carabid beetle assemblage of the experimental burnt site at Ameland (PB1), 6-7 months after burning took place. B= burnt site; C= control site. Abbreviations between brackets: GE= Grass encroached vegetation; Cor=Corynephorus canescens dominated vegetation. Numbers shown at each division are eigenvalues (between 0.0-1.0) associated with that division. Scores associated with carabid species are abundance classes from lowest (1) to highest (9).

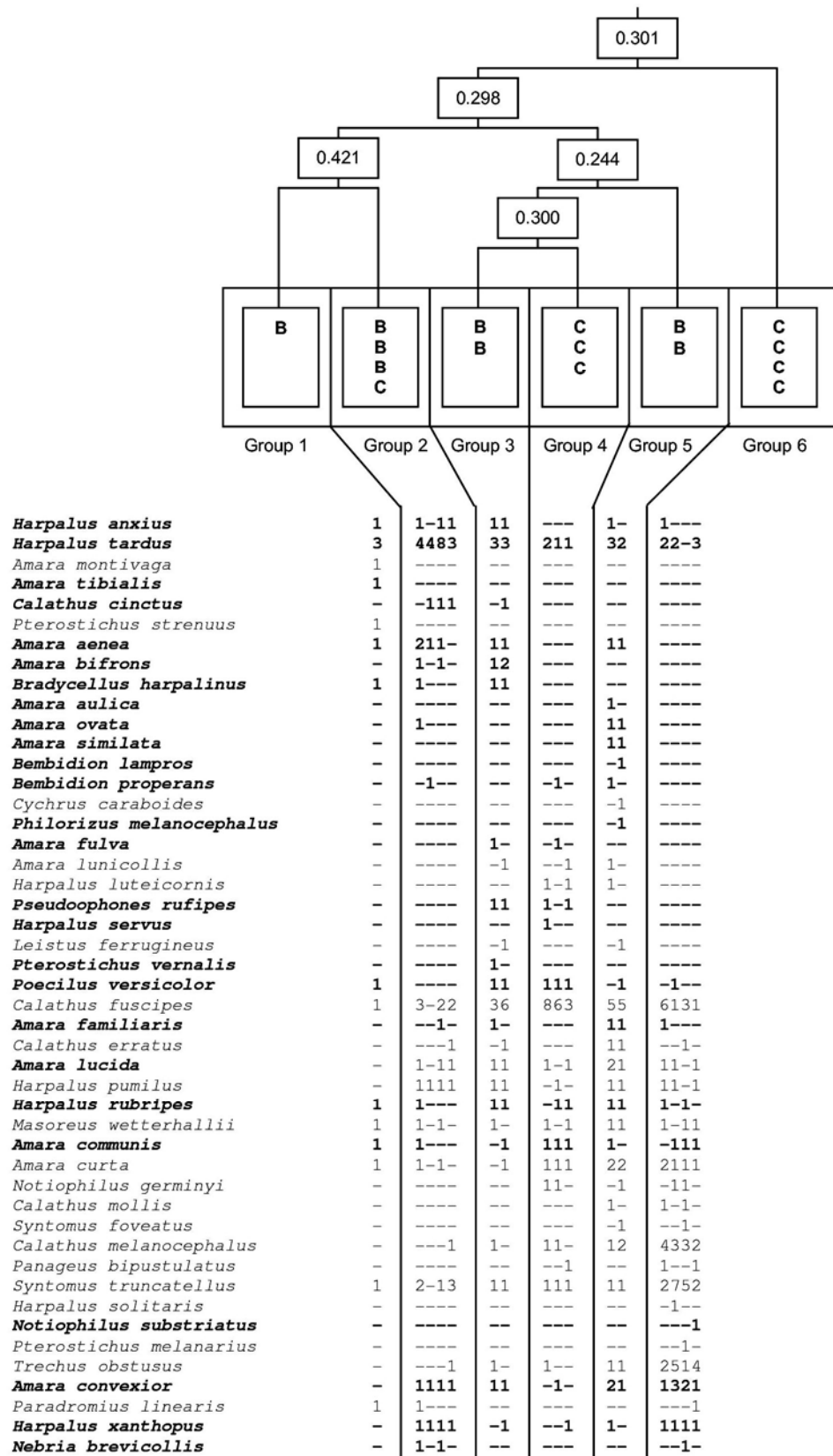
6, consisting of four control sites lacks a substantial number of flight capable species, which are present in the other sites. Differences in species composition between the two groups that were separated in the fifth division (group 4 (burnt sites) and 5 (control sites)) shows the same pattern; several flight capable species that are found in the burnt sites, have not been found in the control sites. The difference in occurrence of flight capable species compared to flight incapable species was also pronounced when considering individual densities: the relative number of species caught with flight capability was significantly higher in the burnt area compared to the unburned control ( $\chi^2$ -test; obs: 208; exp: 154.62;  $p < 0.001$ ).



|                                |   |    |     |    |
|--------------------------------|---|----|-----|----|
| <i>Carabus arvensis</i>        | 3 | -- | 11- | -1 |
| <i>Pterostichus diligens</i>   | 2 | 1- | 11- | -- |
| <i>Demetrias monostigma</i>    | 1 | 12 | 1-- | -- |
| <i>Amara majuscula</i>         | - | -- | -1- | -- |
| <i>Calathus mollis</i>         | - | -- | 1-- | -- |
| <i>Clivina fossor</i>          | - | 11 | -1- | -- |
| <i>Poecilus versicolor</i>     | - | 11 | 121 | -- |
| <i>Paradromius linearis</i>    | - | 12 | --- | -1 |
| <i>Harpalus latus</i>          | - | 32 | -1- | -2 |
| <i>Trechus obstusus</i>        | - | 22 | -21 | -2 |
| <i>Calathus melanocephalus</i> | 2 | 48 | 782 | 89 |
| <i>Oxypselaphus obscurus</i>   | 4 | 54 | 522 | 48 |
| <i>Pterostichus niger</i>      | 2 | 32 | 321 | 22 |
| <i>Trechus quadristriatus</i>  | 1 | 11 | 1-1 | 12 |
| <i>Calathus fuscipes</i>       | - | -- | 221 | 47 |
| <i>Nebria brevicollis</i>      | 1 | -- | --- | 11 |
| <i>Bradycellus harpalinus</i>  | - | -- | --- | -1 |
| <i>Bradycellus ruficollis</i>  | - | -- | --- | 1- |
| <i>Calathus erratus</i>        | - | -- | --- | 31 |
| <i>Notiophilus aquaticus</i>   | - | -- | --- | -1 |
| <i>Syntomus foveatus</i>       | - | -- | --- | -1 |
| <i>Syntomus truncatellus</i>   | - | -- | --- | -1 |

**Fig 4:** Twinspan dendrogram of the carabid beetle assemblage on the wildfire site at Terschelling (WF1) three months after burning. B= burnt sample sites; C=control sample sites. Abbreviations between brackets: Amm= *Ammophila arenaria* dominated; Emp= *Empetrum nigrum* dominated; Cor= *Corynephorus canescens* dominated. Numbers shown at each division are eigenvalues (between 0.0-1.0) associated with that division. Scores associated with carabid species are abundance classes from lowest (1) to highest (9)





**Fig 5:** Twinspan dendrogram of the carabid beetle assemblage at the Castricum wildfire site (WF2). B= burnt sample sites; C=control sample sites. Numbers shown at each division are eigenvalues (between 0.0-1.0) associated with that division. Scores associated with carabid species are abundance classes from lowest (1) to highest (9). Species in bold are capable of flight.

## 4. Discussion

### Burning experiments

In both experimental burning sites, the total biomass removed by the fire was low. For any measure to be successful in terms of nutrient removal, substantial amounts of accumulated litter present in the system have to be removed. This was not the case in both burning experiments; no significant reductions in neither moss nor litter biomass have been recorded.

Possible causes for the low impact of the burning experiments can be found in the method of execution. Both experimental fires were carried out as headfires. The use of headfires results in generally fast-moving fire fronts. As a result, temperatures in the soil remain generally low (Whelan, 1995). Another possible cause for the low removal of biomass was the relative high moisture content of the moss and litter layers. Although the timing of the experimental fires was aimed at a point in which the moisture content of these layers was as minimal as possible, at the time of the execution, water content of moss and litter layers was still high enough to prevent combustion. After six months standing crop biomass of the experimental prescribed burning site was fully recovered. Similar results were found in the species composition of the burnt sites compared to the control sites: the initial minor impact of burning on species composition disappeared within six months after the fire. Furthermore, carabid beetle species composition clearly differed between relative intact dry dune grasslands and the grass-encroached sites. Recovery of the burnt sites towards the species composition of these intact sites did not take place. The impact of the burning experiments on ecosystem functioning could therefore be classified as generally low.

### Wildfires

The wildfire site at Terschelling showed that under the right circumstances, fire is capable in removing substantial amounts of accumulated litter in dune grasslands. This fire took place under much dryer circumstances than the prescribed burning experiments resulting in higher fire intensity and significant reductions in biomass of moss and litter layers. The rate of vegetation recovery depended on the type of vegetation that had been subject to fire. Vegetation recovery in *E. nigrum* dominated vegetation was less marked than in the *A. arenaria* dominated vegetation. This was probably an effect of differences in intensity of the fire between both vegetation types. Fires in *E. nigrum* dominated vegetation are known to reach high temperatures (Vestergaard and Alstrup, 1996). This is probably a result of optimal aeration characteristics of the branched, shrub-like growth form of the vegetation. Possibly, the presence of highly flammable resins present in *Empetrum* play a role in the intensity of the fire as well, although the occurrence of the latter effect still needs confirmation. The post-burn vegetation development was in both vegetation types similar: *A. arenaria* was able to quickly recolonise the area by means of vegetative regeneration; ultimately resulting in the same grass-encroached condition in the case of the *Ammophila* sites and a shift from *Empetrum* dominance towards *Ammophila* dominance in the former *E. nigrum* vegetation. This shift in vegetation dominance in *Empetrum* sites was also found in the investigation of Vestergaard and Alstrup (2001) in Danish dune heaths that were subject to wildfires. The authors argue that this period of dominance by *A. arenaria* will be short-lived as a result of the lack of fresh sand supply, of which *A. arenaria* is dependent for survival (Van der Putten and Van der Stoel, 1998). In the Netherlands, however, both increased nitrogen deposition (Kooijman et al., 1998) and acidification (De Boer et al., 1998) enables *A. arenaria* of gaining and holding a dominant position in stable dunes as well. For example, the occurrence of a wildfire in 1993 on the same island in grass-encroached, former grey dune vegetation resulted in a fast development towards the grass-encroached *Ammophila* dominated state (Ketner-Oostra et al., 2006). The biomass data of the wildfire site in the lime rich dunes near Castricum (WF2), confirm this observation. One year after the fire, standing crop as well as litter biomass fully recovered to unburned conditions; moss biomass however was still lower compared to unburned conditions. In accordance to the developmental processes in the lime poor dunes, vegetation recovery towards tall-grass dominance developed rapidly.

The carabid beetle assemblage of both wildfire sites are considerably different both in total species richness as in species composition. These differences are probably a result of differences in soil chemistry (lime poor versus lime rich dunes), the resulting differential effect of nitrogen deposition and acidification on ecosystem functioning (Kooijman et al., 1998) and ultimately, the degree of degradation of the pre-burn habitat and surroundings. This resulted in a generally species poor carabid community at the Terschelling wildfire site, which harbored a low number of characteristic species of dry dune grassland communities, whereas the species composition of the Castricum

wildfire site was both rich in total number of species, as well as species characteristic for dry dune grassland communities.

As a result of the high intensity of the fire, the carabid beetle assemblage of both wildfire areas showed a rather strong reaction to the fires. In both wildfire sites, the species composition in burnt vegetation differed to unburned controls in the direct vicinity. This difference was at least partially explained by local extinction of several species (which occurred most markedly in the high intensity burned *Empetrum* sites at Terschelling) followed by recolonization of the area from on-site survivors as well as immigrants from the surroundings. Recolonization by means of immigration has been found in the Castricum wildfire site. The occurrence of several flight capable species in the burnt area was largely responsible for the differences in species composition between sampling points in burnt sites and between all sampling points in burnt and unburned sites. Furthermore, the relative number of flight capable species caught in burnt sites compared to control sites was higher than would be expected if recovery was solely driven from local survival. As a result of this colonization, total species richness was higher in the burned sites compared to the unburned nearby vegetation.

However, this effect only occurred at the relatively species rich Castricum site. This suggests that the area surrounding the Castricum burn site, when compared to the Terschelling site, has a higher level of heterogeneity in carabid beetle communities present. Successful recolonization of the area by characteristic species therefore depends on the presence of intact, species rich carabid beetle habitats in the direct vicinity of the burnt area.

## 5. Concluding remarks

One of the difficulties concerning burning management as a restoration tool involves the method of burning. This study shows that at present, experience and knowledge on how the technique of burning is to be used is largely lacking. Therefore, it remains questionable if the use of winter fires is a viable management option as a tool to remove substantial amounts of nutrients. The results of the wildfires show that burning can remove substantial amounts of organic material, although whether or not the same results can be obtained with a controlled fire remains an open question.

Furthermore, it is clear that burning alone is not capable of restoring dry dune grassland vegetation. Fast vegetative regrowth of tallgrasses quickly results in the re-establishment of grass-encroached vegetation. In the worst scenario, the end-result is even less favourable than the previous condition (as observed in the rapid post burn grass-encroachment of *Empetrum* dune heaths). The use of additional management aimed at the inhibition of fast vegetative regrowth of tallgrasses could be a viable option to reduce this effect. The most obvious management option is the use of grazing following burning. Several studies have shown a preference of freshly burnt area's by large herbivores (Hobbs et al., 1991; Vandvik et al., 2005; Anderson et al., 2006), resulting in a high grazing pressure on the target area, which in this case could possibly result in the successful restoration of dry dune grasslands. The recovery of fauna communities characteristic of dry dune grasslands provides us an even greater challenge. Successful restoration of fauna communities depends on several factors, at least including the presence of nearby relict populations, survival –immigration interactions and the suitability of the post-burn habitat (Anderson et al., 1989; Delettre, 1994; Swengel, 2001; Panzer, 2002; Hochkirch and Adorf, 2007).

In order to improve the level of knowledge involving burning management to both nature managers and researchers in the Netherlands, the Ministry of Agriculture, Nature management and Food quality has commissioned a literature study on the effects of prescribed burning in nature areas. This study is nearly completed and the report (Bobbink et al., 2009) will be soon available. Topics addressed by this literature study are: the effects of prescribed burning on (i) physio-chemical soil properties, (ii) the nutrient-state and nutrient availability and (iii) biota (including flora, fungi and fauna). Environmental parameters (type of vegetation; weather conditions, etc.) that influence the outcome and effectiveness of burning-management and the type of burning technique used on the outcome of the measure are also included in this literature study

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