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# Managing Invasive Alien Plants in Ireland

Authors: Vasiliki Balogianni, Bruce Osborne,
Margherita Gioria and Dario Fornara

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Rialtas na hÉireann
Government of Ireland

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- Office of Communications and Corporate Services

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**EPA RESEARCH PROGRAMME 2021–2030**

# Managing Invasive Alien Plants in Ireland

## (2016-NC-MS-7)

## EPA Research Report

Prepared for the Environmental Protection Agency

by

UCD School of Biology and Environmental Science, and UCD Earth Institute,
University College Dublin

**Authors:**

**Vasiliki Balogianni, Bruce Osborne, Margherita Gioria and Dario Fornara**

**ENVIRONMENTAL PROTECTION AGENCY**
An Ghníomhaireacht um Chaomhnú Comhshaoil
PO Box 3000, Johnstown Castle, Co. Wexford, Ireland

Telephone: +353 53 916 0600  Fax: +353 53 916 0699
Email: info@epa.ie  Website: www.epa.ie

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**Cover image: (a)** A grassland community invaded by Himalayan balsam, *Impatiens glandulifera*, close to the banks of the River Lagan, Belfast. The adjacent uninvaded community was dominated by *Urtica dioica* and contained individuals of giant hogweed, *Heracleum mantegazzianum*, which had spread from the riverbank nearby. **(b)** A coastal grassland community invaded by *Gunnera tinctoria* on Achill Island, Co. Mayo, at the beginning of the growing season in April. (Photos by Margherita Gioria.)

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The EPA Research Programme addresses the need for research in Ireland to inform policymakers and other stakeholders on a range of questions in relation to environmental protection. These reports are intended as contributions to the necessary debate on the protection of the environment.

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# Project Partners

**Bruce Osborne**
UCD School of Biology and Environmental Science
and
UCD Earth Institute
University College Dublin
Belfield
Dublin 4
Ireland
Tel.: +353 1 716 2249
Email: Bruce.osborne@ucdconnect.ie

**Margherita Gioria**
UCD School of Biology and Environmental Science
and
UCD Earth Institute
University College Dublin
Belfield
Dublin 4
Ireland
Email: margherita.gioria@ibot.cas.cz

**Vasiliki Balogianni**
UCD School of Biology and Environmental Science
and
UCD Earth Institute
University College Dublin
Belfield
Dublin 4
Ireland
Email: vasiliki.balogianni@gmail.com

**Dario Fornara**
Sustainable Agri-Food Sciences Division
Agri-Food and Biosciences Institute
Belfast
UK
Tel.: +44 2890 255 335
Email: Dario.fornara@afbini.gov.uk

# Other Project Contributors

**Laura Gallego**
UCD School of Agriculture and Food Sciences
University College Dublin
Belfield
Dublin 4
Ireland
Email: laura.gallego-lorenzo@ucd.ie

**Amanuel Gebremichael**
Teagasc Research
Johnstown Castle
Wexford
Ireland
Email: amanuel.gebremichael@teagasc.ie

**John McIlroy**
Sustainable Agri-Food Sciences Division
Agri-Food and Biosciences Institute
Belfast
UK
Email: john.mcilroy@afbini.gov.uk

**Tony Gamelin**
Établissements publics locaux d’enseignement et de formation professionnelle agricoles (EPLEFPA)
Bordeaux-Gironde
France
Email: tony_gamelin@yahoo.fr

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# Executive Summary

This report is aimed at providing a theoretical background to inform the control of major terrestrial invasive plants in Ireland, based on their community and ecosystem impacts, their ecology and their competitive interactions with native species. The research evaluated current knowledge on the impact of terrestrial invasive alien plants and their management in Ireland; it also examined the effects of two large herbaceous invasive plants – *Gunnera tinctoria* (giant rhubarb or Chilean rhubarb) and *Impatiens glandulifera* (Himalayan balsam) – at the community and ecosystem levels, and assessed their soil seed bank characteristics and competitive ability. The study species were selected because of their contrasting ecology and physiology, their status as high-impact invasive species in Ireland and their classification as species of Union concern according to the recent EU Regulation on Invasive Alien Species (Regulation No. 1143/2014). Based on the results, recommendations for the development of management, monitoring and control measures are also provided.

The research detected a lack of quantitative information on alien invasive species impacts in Ireland, and while most studies observed decreased plant diversity in invaded communities, this was limited to short-term above-ground assessments. This study highlights the need for a significant investment in long-term research if we are to protect Ireland's biodiversity and ecosystems, with a focus on how plant invasions influence ecosystem functioning and the provision of essential ecosystem services.

The research showed community and ecosystem impacts to be clearly invader dependent and thus the need to take into account the specific identity of the invading species for management purposes cannot be overemphasised. While *G. tinctoria* had a major negative impact on plant communities, both above and below ground, *I. glandulifera* had a more diverse effect, strongly depending on the site. Although both the species investigated had significant impacts at the ecosystem level, these differed both qualitatively and quantitatively. Local environmental factors and management history may also interact with species-specific traits, indicating that there is a complex matrix of biotic and abiotic interactions that influence ecosystem impacts.

Examining competitive interactions between *G. tinctoria* and co-occurring native species has led the researchers to challenge two widespread notions. The first was that invasive species are always strong competitors, as *G. tinctoria* proved to be a poor competitor. This was because its productivity was greatly reduced and the probability of the survival of young seedlings was very low, even under the lowest competitive pressure used. Early colonisation and establishment is still likely to depend on factors other than a higher competitive ability, such as its highly asynchronous germination. In later stages of *G. tinctoria* invasion, traits such as vegetative propagation, high seed production and, for mature plants, its great size, which would result in the shading out of many co-occurring shade-intolerant species, might compensate to some extent for competitive limitations. In terms of the mechanism underlying its poor competitive ability, the evidence suggests that a limited capacity to utilise soil phosphorus may be the more important factor.

The second notion was that *G. tinctoria* is always reliant on nitrogen fixation. In the fertilised treatments, where *G. tinctoria* was growing alone, the study found an increased nitrogen concentration in both above- and below-ground *G. tinctoria* tissues, while the nitrogen isotopic ratio ($\delta^{15}N$) showed greater discrimination (positive values) than for the unfertilised treatments. As the $\delta^{15}N$ value of plants that rely exclusively on atmospheric nitrogen is usually close to zero or slightly negative, this could indicate that *G. tinctoria* might have some capacity to utilise soil mineral nitrogen when higher concentrations of nitrogen are available. Although the isotopic signature can be confounded by internal nitrogen remobilisation and nitrogen transformations, this may challenge the notion that *G. tinctoria* depends exclusively on nitrogen fixation to meet its nitrogen requirements. If *G. tinctoria* can utilise soil nitrogen when available in excess in the absence of competition, then it will be able to both survive in nitrogen-depleted environments through nitrogen fixation and thrive in overfertilised environments through complementary nitrogen uptake.

Finally, the results were evaluated in relation to current management strategies used in Ireland for the two species studied and used to provide recommendations for future control measures. Although the focus of most management practices has been on the application of herbicide treatments and/or mechanical removal, there is a clear need for alternative, site- and invader-specific, approaches that account for the long-term impacts of plant invaders and their contrasting ecologies.

# 1 Introduction

## 1.1 Objectives

This project was aimed at providing the theoretical background for developing control and eradication measures for two major invasive alien plants in Ireland, based on an understanding of their ecology (including their competitive ability and phenology), their impacts on communities and ecosystems, and the characteristics of the invaded communities. It relied on a combination of different approaches that have been advocated to provide promising avenues for the long-term control of invasive plants (Grman and Suding, 2010; Wolkovich and Cleland, 2011; Wainwright and Cleland, 2013; Godoy and Levine, 2014; Passos *et al*., 2017; Nsikani *et al*., 2018).

The specific objectives of the project were:

- to examine the impact of two major invasive species in Ireland at the community level (above- and below-ground vegetation, including the seed bank) and ecosystem level (alterations in nutrient cycling);
- to compare the characteristics of native species that must be accounted for in developing sustainable control and restoration measures, such as competitive ability, germination phenology and soil seed bank characteristics, in different ecosystems/habitat types in Ireland;
- to develop control measures that account for the characteristics of invasive alien species, of native species and of the invaded communities, and which account for the modifications that have been caused by plant invasions, including changes in the above-ground vegetation, long-term effects on the soil seed bank and alterations in nutrient cycling;
- to provide recommendations for the development of management and restoration measures for invasive alien (terrestrial) plants in Ireland and for similar habitats, and environmental conditions, in other regions of the world.

## 1.2 Work Packages

The study objectives were addressed using a range of experimental and theoretical approaches in five work packages (WPs).

### *1.2.1 Impact assessments and management strategies for invasive alien plants in Ireland (WP1)*

According to EU Regulation 1143/2014 (EU, 2014) on invasive alien plants (IAS) – the IAS Regulation –, invasive alien plants are plants introduced outside their natural range and whose introduction or spread has been found to threaten or adversely impact biodiversity and related ecosystem services. Ireland is highly susceptible to invasions by alien plants with broad ecological ranges, probably in large part because of its relatively mild climate (Gioria and Osborne, 2010). This is reflected in the fact that alien plant species account for a large proportion of the total number of species present (900 native species vs 600 alien/naturalised species; Reynolds, 2002). In Ireland, many invasive plant species have become well established, such as the shrubs *Rhododendron ponticum* (rhododendron) and *Prunus laurocerasus* (cherry laurel), and the large herbs *Fallopia japonica* (Japanese knotweed), *Gunnera tinctoria* (giant rhubarb or Chilean rhubarb) and *Impatiens glandulifer*a (Himalayan balsam) (Reynolds, 2002; Stokes *et al*., 2006; Gioria and Osborne, 2010, 2013). These have colonised a variety of ecosystem types, with substantial impacts on Ireland's biodiversity and ecosystems. The widespread *Heracleum mantegazzianum* (giant hogweed) also has health implications, as contact with furanocoumarins in its sap causes photodermatitis, manifesting as mild to severe erythematous reactions (red colouring of the skin) with or without painful blisters, depending on the quantities of sap and exposure to ultraviolet (UV) rays (Thiele and Otte, 2007). The objectives of this WP were:

1. to review the state of research on the impacts of invasive alien plants on terrestrial ecosystems in Ireland, including changes in biodiversity, ecosystem functioning and services;
2. to describe the measures that have been put into place to manage and eradicate high-impact terrestrial invasive alien plants in Ireland, including the costs of such measures and whether or not they have been successful in Ireland and globally;

3. to identify the research gaps, management challenges and the measures that need to be put into place to comply with the new EU regulations on invasive alien species.

### *1.2.2 Impact of major invasive alien plants on native communities (WP2)*

Developing successful control and restoration measures must be based on a comprehensive understanding of the ecology and biology of invasive and resident (natives and other alien) species in a community and of the impact(s) of plant invasions at the community and ecosystem levels. To date, however, the control and eradication measures used have mainly been uncoordinated and have predominantly been directed at the short-term *ad hoc* implementation of mechanical removal and/or herbicide application, focused almost exclusively on the rapid reduction/removal of the above-ground biomass of the target species. It is perhaps not surprising that this has, in many cases, had limited success. Such approaches take little account of differences in the characteristics of the target species, or the environments that they invade, as they require species- and site-specific management interventions; however, there is the strong possibility that these actions might be counterproductive and promote the germination of a large number of propagules of the invader and/or other unwanted species (e.g. Gioria and Osborne, 2010).

Recent studies have shown that sustainable management measures, which minimise environmental, social and economic costs while restoring resilience to ecosystems and creating robust social and economic supports for the implementation of management plans (Larson *et al*., 2011), require knowledge of (1) the soil seed bank of the invasive species and (2) an understanding of the range and magnitude of any ecosystem impacts, including the effects on the soil seed bank of invaded communities (Gioria and Osborne, 2010; Wolkovich and Cleland, 2011; Gioria *et al*., 2012; Cordell *et al*., 2016; Nuñez *et al*., 2017). Despite the importance of soil seed bank and ecosystem impacts, most studies have focused on an examination of the effect of alien plants on the above-ground vegetation. While such a focus has demonstrated that invasions often result in substantial decreases in species richness and the formation of virtually monospecific stands (Hejda *et al*., 2009; Powell *et al*., 2013), a multitude of other effects that will be important for both control and restoration purposes have often been ignored (Vilà and Gimeno, 2007; Gioria *et al*., 2014). The impact on the soil seed bank (hereafter seed bank), which largely governs the ability of ecosystems to recover after the invader has been removed, has received comparatively less attention (Gioria *et al*., 2012), probably because of the practical difficulties associated with seed bank assessments (Gioria and Osborne, 2009a). Clearly, this is a key issue in any successful eradication or management programme, and for restoration purposes.

Seed banks are reserves of seeds in the soil or on its surface, composed of dormant and/or non-dormant seeds, representing a memory of past and present vegetation (Harper, 1977). The functional role of seed banks in plant invasions has been reviewed by Gioria *et al*. (2012). Seed banks are a major determinant of vegetation dynamics and represent a mechanism for species coexistence associated with the differential response of the species in a community to changing environmental conditions and/or use of resources. The formation of a seed bank by invasive plants will act as a source of propagules and genetic diversity, providing them with an improved capacity to respond to the novel conditions encountered in their alien range and enhancing their invasive potential (Gioria *et al*., 2014). An important distinction is that between transient (persisting in the soil for less than 1 year), short-term persistent (between 1 and 5 years) and long-term persistent (over 5 years) seeds (Thompson *et al*., 1997). This classification is very important in the context of plant invasions, as it allows us to predict for how long invasive (and native) species may persist in the soil in the absence of further introductions. This information is indispensable for developing sustainable control programmes and for assessing the potential role of native seed banks in the revegetation of previously invaded areas (Gioria *et al*., 2012).

The legacy of plant invasions and the success of management and restoration measures will ultimately depend on the persistence of an invasive species in a community and on the capacity of recipient communities to resist the full or partial displacement of resident species from the above-ground vegetation (Gioria *et al*., 2011, 2012). These factors are strongly linked to the seed bank of invaded communities.

The soil seed bank study was focused on quantifying the number of viable propagules in the soil to assess if eradication is a practical option. Furthermore, these studies provided information related to the type of seed bank formed by the invasive species (transient, short-term persistent or long-term persistent; *sensu* Thompson *et al.*, 1997), which is crucial for assessing how long it would take to deplete the seed bank of an invasive species, which in turn determines if long-term control would be effective. In addition to examining invasive species, the study also investigated possible losses of native species from the soil seed bank and provided information on the susceptibility of native communities to plant invasions and the susceptibility of invaded communities to secondary invasions by other alien species.

### *1.2.3 Impact of major invasive alien plants on ecosystems (WP3)*

The legacy of plant invasions is also reflected in their long-term effects on invaded ecosystems, such as changes in nutrient cycling or hydrology and their influence on soil biota, pollinators and/or predators. Changes in nutrient cycling or increases in soil nitrogen through increased litter production or the influence of nitrogen-fixing invasive species may have major implications for the management of invasive plants and the restoration of native communities. Nitrogen enrichment, for instance, could facilitate the establishment of nitrophilous weedy species, such as *Urtica dioica* (common nettle) and *Galium aparine* (cleavers), or alien species that tend to colonise nitrophilous communities, such as *Erigeron sumatrensis* (Guernsey fleabane), while decreasing the abundance of native species, including rare species that typically have low resource requirements.

Invasive plants may alter the functions of, and services provided by, native ecosystems through a range of mechanisms associated with changes in species composition, including species displacement. Often invasive species tend to form near-monospecific stands, where plant species diversity is greatly reduced (Figure 1.1). These invaded plant communities are usually associated with higher above- and below-ground biomass production than the species they replace and thus produce more plant litter, altering the quality and quantity of decomposing material and the consequent release of nutrients, with potential effects on biogeochemistry and the quality and quantity of light reaching the soil, and modifying the microclimatic conditions (D’Antonio and Corbin, 2003; Dehlin *et al.*, 2008; Bansal *et al.*, 2014). Increases in nutrient concentrations in surface soil layers of fertile sites may result in nutrient losses and modifications in soil chemical, physical and biological properties, which may, in turn, affect rates of soil organic matter formation and turnover.

Moreover, certain invasive species, including *G. tinctoria* (Osborne *et al.*, 1992; Mantoani *et al.*, 2020), are capable of fixing atmospheric nitrogen and could increase the level of nutrients available to other plants, either directly or through the sparing use of soil

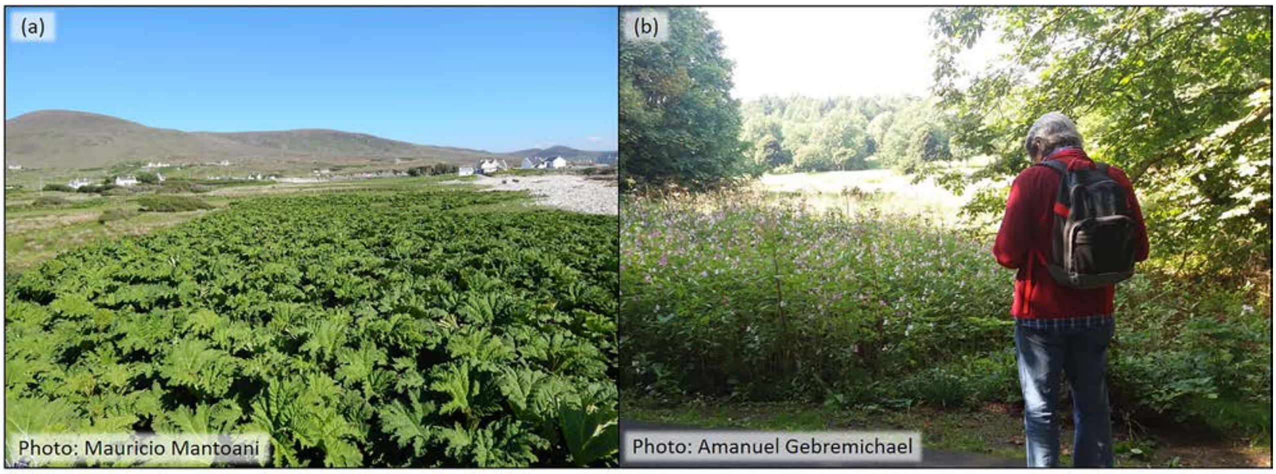

**Figure 1.1. Largely monospecific stands of (a) *G. tinctoria* invading a coastal grassland on Achill Island, Co. Mayo, Ireland, with adjacent native vegetation on the left, and (b) *I. glandulifera* invading the banks of the River Lagan in Lagan Valley Regional Park, Belfast, UK.**

reserves. Changes in the environmental conditions in invaded communities may affect each stage of plant development, from the seed stage, with seeds that may become more susceptible to the effects of pathogens and predators, to lowering recruitment from the seed bank or the survival rate of seedlings or even by affecting the successful establishment and growth of seedlings. Furthermore, the susceptibility of a plant community to colonisation and dominance by introduced organisms, often referred to as its invasibility (Fridley, 2011; Guo *et al*., 2015), is thought to be increased by the intermittent increase in resources in terrestrial habitats (Davis *et al*., 2000); thus, the increased nutrient availability resulting from invasions might increase the potential for secondary invasions. Knowledge of changes in nutrient availability due to the impacts of invasive species is essential to develop restoration measures that are based on the environmental conditions prevailing at a given location.

Despite the high potential for alien plant species to affect the biogeochemistry of invaded areas, surprisingly little information is available on their potential impact on critical ecosystem services, such as soil nutrient availability and the accumulation of carbon in soils (i.e. soil carbon sequestration). These two services are key supporting services (see the Millennium Ecosystem Assessment, https://www.millenniumassessment.org) on which other provisioning, regulating and cultural services all depend. Highly diverse plant communities contribute to the delivery of these ecosystem services because soil resources are used more efficiently by a different array of plant species. The main question remains if and how a reduction in biodiversity in invaded plant communities could affect the delivery of these two ecosystem services, both locally and at larger spatial scales (e.g. at the river catchment scale). To answer this question, we need to search for potential linkages among (1) the invaded community, (2) soil nutrient cycling and (3) the accumulation or loss of soil organic matter (carbon, nitrogen and phosphorus), and compare these with uninvaded plant communities. Very few studies so far have addressed such linkages, partly because of the interactive effects of many environmental variables (e.g. climate, soil type, identity of the invasive species, structure and composition of the native community, and disturbance history). This means that long-term ecosystem responses to invasion are often site-specific and require a multidisciplinary approach when multiple above- and below-ground variables are simultaneously measured.

The main objective of this WP was to address the potential ecosystem-level impact of invasive plant species by assessing the soil legacy and community-level impacts of plant invasions at sites in Ireland and Northern Ireland in parallel WPs.

### 1.2.4 Comparing the ecology and phenology of invasive alien and native species for control and restoration purposes (WP4)

Recent studies show that sustainable control measures require knowledge of the timing of germination of seeds of invasive and alien species (germination phenology) (Gioria and Osborne, 2017; Gioria and Pyšek, 2017; Gioria *et al*., 2018a). Additional factors that may play a key role in the success of control (and restoration) measures include the competitive ability of invasive and native species and phenological differences between invasive and native species.

The competitive ability of a species can be defined as its ability to exploit limiting resources and can also be defined as the rate of resource consumption (see Gioria and Osborne, 2014). It also reflects the capacity of a species to reduce the resources available to other neighbouring plants (maximum resource capture) or its ability to tolerate low-resource conditions (minimum resource requirement) and has long been considered a key factor in the successful establishment of several invasive species (Elton, 1958; Tilman, 1997; Grime, 2001). In particular, the detrimental effects of many individual invasive species, including a reduction in biodiversity, homogenisation of the native biota and the formation of near-monospecific stands (e.g. McKinney and Lockwood, 1999; Gioria and Osborne, 2010), have often been associated with the fact that their capacity to compete for resources is superior to that of native species (for a review, see Gioria and Osborne, 2014). The role of resource competition in determining naturalisation and invasion by alien species has largely been inferred from the higher productivity that characterises many invasive species, particularly under increasing levels of resources, while few studies have examined resource competition directly. Determining if resource competition plays a major role in the success of an alien species and identifying under which environmental conditions invasive species are competitively superior to native species is essential for developing management and restoration measures. Moreover, invasive species may benefit disproportionately from increases in resources compared with native species (Gioria and Osborne, 2014). If an alien plant is competitively superior to native species, its control could involve altering the resources in a way that increases the competitive ability of native species. In many cases, native plant communities are competitively superior to alien species in conditions of low resource availability (Vasquez *et al*., 2008; Going *et al*., 2009), but cases of increases in resources resulting in improved competitive ability of native species have also been recorded (Going *et al*., 2009; Balogianni *et al*., 2017). In some situations the competitive ability of native species might not even be affected by changes in resources (Kolb and Alpert, 2003).

Knowledge of the competitive ability of invasive and native plants, under varying environmental conditions, should also be considered in control programmes as a basis for identifying ways of suppressing the growth and development of the introduced species. Depending on the identity of the invader, these measures could include increases in the availability of one or more limiting resources to increase the competitive ability of native species, and thus their capacity to suppress the growth of invasive plants, or reductions in resource supply, which could favour native species that are adapted to low levels of one or more resources.

Plant phenology, i.e. the timing of phenological events, such as the onset of seasonal growth, or the timing of reproduction or dispersal, is a key trait determining the way in which plant communities are assembled (Wolkovich and Cleland, 2011). Phenological niches arising from differences in the timing of phenological events may create windows of opportunity that could favour invasive species by initiating growth or germination earlier/later than native species (Engelhardt and Anderson, 2011; Wolkovich and Cleland, 2011; Gioria *et al*., 2018a; Alexander and Levine, 2019). The effects associated with the early capture of resources, faster growth and a higher probability of survival are often called priority effects. Through these priority effects a competitive advantage may be gained by allowing the species with the earlier phenology to uptake and utilise resources and space before other species are active. The early utilisation and sequestration of resources by early-arriving species may lead to a greater seasonal growth and a disproportionate share of available resources, thus significantly suppressing the growth of smaller species, which is a phenomenon called size-asymmetric competition (Schwinning and Weiner, 1998). In addition to the pre-emptive use of resources, the earlier arriving, larger species might also have significant shading effects on their competing neighbours (e.g. Gioria and Osborne, 2009a,b).

Priority effects originating through the earlier emergence of alien plants have been observed for many species in several ecosystem types (e.g. Wolkovich and Cleland, 2011; Gioria and Osborne, 2010, 2013). Early germination and development by an alien species may have a large impact on competitive interactions with native species (Wolkovich and Cleland, 2011). By emerging earlier than native species, an alien species may avoid competition for some resources at the initial stage of establishment, and the relative importance of competition for resources may be reduced. Seasonal priority effects may allow exotic species to fill a vacant phenological niche by growing earlier in the season than native species (Wolkovich and Cleland, 2011). Early-emerging species may also affect the establishment of later-emerging species through changes in the soil conditions (Grman and Suding, 2010), thus exerting control over native species that emerge later in the season. Early germination by alien introductions may be particularly detrimental to light-demanding species that achieve maximum carbon gain/cover before canopy closure, such as spring ephemerals, or in communities of slow-growing species or forests (Gioria and Osborne, 2010).

Identifying and quantifying priority effects is important for the development of control and restoration strategies, as potential priority effects may hamper the success of control measures. Control programmes must be designed to suppress the growth of invasive species and reduce any inhibitory effects on the growth and germination of native species. The manipulation of priority effects to suppress the growth of alien species that would otherwise emerge earlier than native species has been recommended as a management technique to restore native communities, especially for less competitive species (Grman and Suding, 2010). Priority effects may be manipulated by sowing seeds of native species that possess a similar germination phenology to invasive species and by manipulating resource availability (e.g. Young *et al*., 2015).

Both differences in competitive ability and in phenology will also affect the invasibility of native communities, i.e. their susceptibility to invasions by alien species. Changes in the seed bank and in soil nutrients associated with plant invasions may also promote secondary invasions by alien species (see Gioria *et al*., 2011). While rarely examined, secondary invasions represent a major additional threat associated with programmes aimed at controlling an invasive species, as the primary invader can create conditions favourable for secondary alien invasions (Gioria *et al*., 2011). The objectives of this WP were, therefore, (1) to compare the competitive ability of invasive and native plant species, (2) to compare the germination and leaf-out phenology of invasive and native plant species and (3) to identify potential priority effects.

### 1.2.5 Developing management and restoration measures (WP5)

Successful plant invasions occur when species that are introduced outside their native range (alien species) form self-sustaining populations (naturalisation phase) and spread at considerable distances from the location of introduction (invasion phase) (Richardson *et al*., 2000). Invasions by alien plants represent a major component of global change, often resulting in dramatic changes in biodiversity and ecosystem functioning (Kumschick *et al*., 2015). This can also be associated with substantial economic impacts (as a result of the loss of ecosystem services and the costs associated with control, eradication, management and restoration measures) (van Wilgen *et al*., 2001). Understanding the mechanisms and processes underlying successful plant invasions has thus become one of the most pressing ecological concerns, given that they are expected to increase and interact with other global changes, such as climate change, nitrogen deposition and increases in atmospheric $CO_2$ (Dukes and Mooney, 1999), and with increased global trade and horticultural activities (Bradley *et al*., 2012).

The need to halt the loss of biodiversity in Ireland, protect habitats of high conservation value and prevent the loss of ecosystem functions and the services they provide requires the development of management, control and/or eradication measures that are cost-effective and sustainable over time, with minimal environmental impact. Where invasive species have had a significant impact at the species, community and/or ecosystem levels, restoration programmes should be developed to restore the functions provided by such ecosystems and, where possible, reintroduce key species that might have been lost. The objectives of this WP were to inform the development of management measures to control the spread of and eradicate, where feasible, the two studied invasive plants and to provide guidelines for the development of restoration measures that are species and habitat specific.

# 2 Methods

WP1 was a desk study reviewing previously published research on the impact of plant invasions in Ireland and elsewhere, where relevant, which focused on control and eradication measures that were used previously, regardless of their effectiveness. WP1 also included an assessment of the cost-effectiveness of such measures. WP2 and WP3 characterised the impacts of invasive plants and provided information on the ecology of invasive and native plants and on the characteristics of invaded communities. WP4 investigated the role of competition and priority effects on invasive species success. WP5 combined the results of all other work packages to propose cost-effective control measures, in conjunction with information based on previous experience.

## 2.1 Work Package 1

WP1 was conducted in the form of a desk study. In addition to reviewing published studies, this WP involved contacting county councils, the National Parks and Wildlife Service, Inland Fisheries Ireland, environmental officers, park managers and relevant community groups and volunteer initiatives addressing invasive species such as *F. japonica* and *I. glandulifera*.

The output from this WP was organised into two review papers (Gioria *et al*., 2018b, 2019). One review dealt with impacts and the other with the management of invasions by terrestrial alien plants in Ireland. The researchers also generated new quantitative information on how the distribution of five high-impact invasive plant species has changed over time, based on changes in the number of recorded invasive alien plants, using data provided by the National Biodiversity Data Centre. These data were generated for the four terrestrial invasive alien plants that are included in the list of species of Union concern (IAS Regulation) and which are present in Ireland (*H. mantegazzianum, G. tinctoria, I. glandulifera* and *Lysichiton americanus*); data were also generated for *R. ponticum*, because of its increasing distribution and impacts in areas of high conservation value. This information can be used to understand to what extent previous management efforts at the national and local levels have been successful at controlling the major invasive alien plants in Ireland; the information also provides a basis for estimating the efforts that would be required to effectively control or eradicate these species.

## 2.2 Work Package 2

### *2.2.1 Above- and below-ground community impact*

This study assessed the impact, both above and below ground, of two large herbaceous invasive species in Ireland – the perennial *G. tinctoria* and the annual *I. glandulifera* – at the species and community levels (Figure 2.1). The study species were selected because of their status as high-impact invasive species in Ireland and elsewhere and because of their classification (since August 2017) as species of Union concern, according to the recent EU Regulation on Invasive Alien Species (IAS Regulation) (Regulation No. 1143/2014). To assess the impacts of *G. tinctoria* and *I. glandulifera* on the vegetation and the soil seed bank, and on soil nutrients (WP3), six suitable sites were identified: three sites for *G. tinctoria*, located on Achill Island, County Mayo, and three sites for *I. glandulifera*, located in Lagan Valley Regional Park, along the River Lagan, County Down, Northern Ireland (for the site location and a description see Figure 2.2 and Table 2.1). These individual sites were comparable in terms of soil type, light and disturbance regime and were representative of the main habitats where these species are invasive in Ireland (coastal grasslands for *G. tinctoria* and riparian habitats for *I. glandulifera*), although these species have also been reported in other habitat types in Ireland. According to Botanical Society of Britain and Ireland (BSBI, n.d.) plant distribution information/maps, *G. tinctoria* was first recorded in the greater study area of Achill Sound during the 1950s (https://bsbi.org/maps?taxonid=2cd4p9h.fbw) and *I. glandulifera* was first recorded in Lagan Valley Regional Park, also during the 1950s (https://bsbi.org/maps?taxonid=2cd4p9h.18x). These dates for *G. tinctoria* are consistent with the soil core dating study of Fennell *et al*. (2012), which indicated that this

**Figure 2.1. The two study species examined in WP2: (a) *G. tinctoria* (Chilean rhubarb) (photo by Mauricio Mantoani) and (b) *I. glandulifera* (Himalayan balsam) (photo by Udo Schmidt, https://creativecommons.org/licenses/by-sa/2.0/legalcode).**

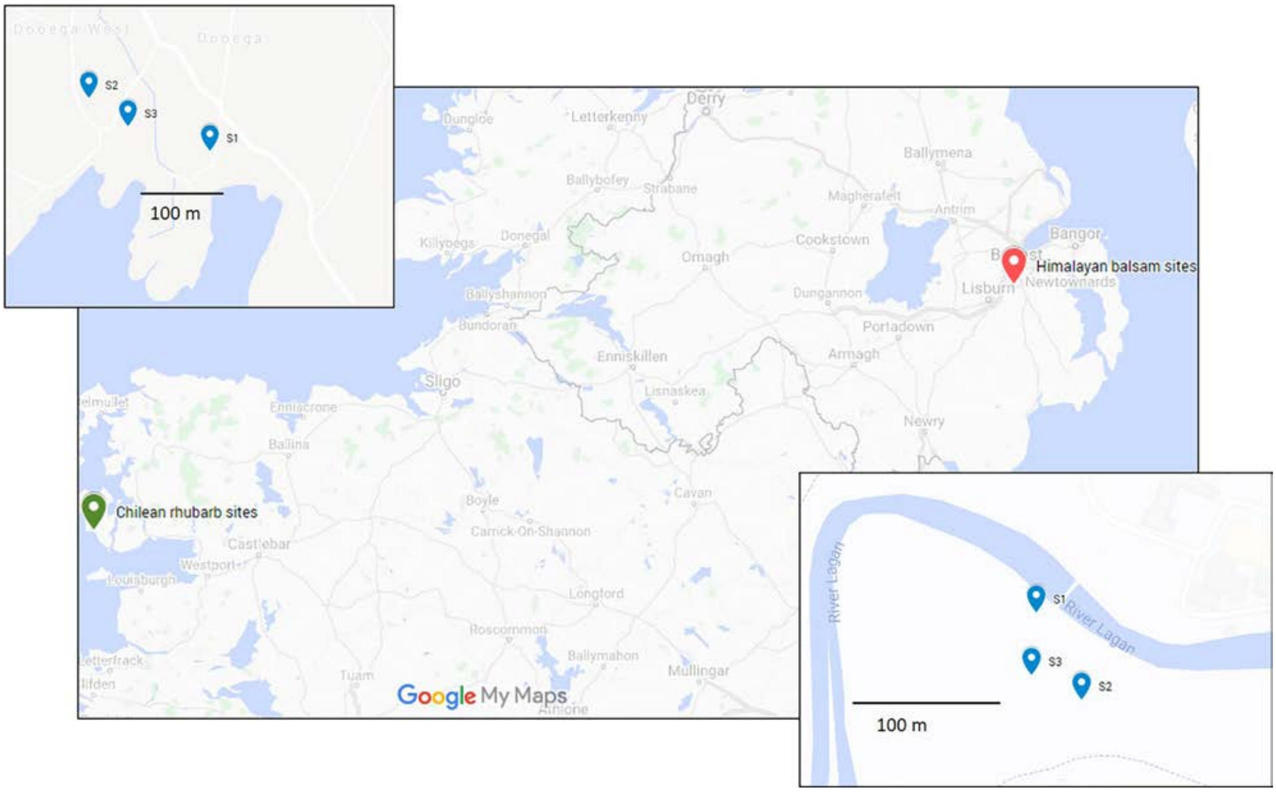

**Figure 2.2. Location of the study sites on the island of Ireland. Map inserts are close-ups of sites associated with *G. tinctoria* on Achill Island, Co. Mayo (top left), and *I. glandulifera*, Lagan Valley Regional Park, Belfast, Northern Ireland (bottom right).**

species may have been a significant presence at the coastal site used in this study for more than 70 years. Our study assessed the impact of invasive alien plants on species diversity, species abundance/density and the species composition of native communities in (1) the above-ground vegetation and (2) the below-ground flora (soil seed bank).

Changes in the vegetation were visually assessed every 4–6 weeks throughout the growing season, in the summer of 2017 and in spring 2018. At each site,

**Table 2.1. Description of the two study areas**

| | **Invasive species** | |
|---|---|---|
| | ***G. tinctoria*** | ***I. glandulifera*** |
| Location | Achill Island, Co. Mayo | Lagan Valley Regional Park, Belfast |
| Geographical co-ordinates | | |
| Site 1 | 53.92168, –10.021578 | 54.554755, –5.949537 |
| Site 2 | 53.92236, –10.025462 | 54.554226, –5.949029 |
| Site 3 | 53.92314, –10.027307 | 54.554389, –5.949600 |
| Dominant plant species[a] | | |
| Invaded sites | *G. tinctoria, U. dioica, G. aparine* | *I. glandulifera, Ranunculus repens, Agrostis stolonifera* |
| Non-invaded sites | *Potentilla anserina, Ranunculus repens, Lolium perenne* | *U. dioica, A. stolonifera, H. mantegazzianum* |
| Habitat type | Coastal grassland | Riparian grassland and shrubland |
| Soil | Shallow mineral and peaty gleysols | Cambisols |
| Land use | Agriculture (sheep and cow grazing); some parts may have been used earlier as cropland | Seminatural, recreational |

**[a]Dominant species varied with site. For this table, species abundance was averaged across the three sites to determine the dominant species in the study area.**

data on the standing vegetation were collected in five comparable invaded plots and five non-invaded plots. Species richness, abundance, dominance and evenness indexes (Shannon diversity index, Simpson's diversity index, Pielou's evenness index) were calculated. Multivariate analyses (permutational multivariate analysis of variance – PERMANOVA, non-multidimensional scaling; see Gioria and Osborne, 2009) were performed on abundance data (cover) (species × abundance matrices).

Assessments of the type and size of the seed bank of invasive and native species were made using a sampling and analytical protocol that was developed specifically to assess changes in the seed bank associated with plant invasions by Gioria and Osborne (2009a). In total, 1200 soil samples were collected from the same sites where the impact on the standing vegetation had been assessed at two points in time: in March 2018, before the germination of most species had occurred in the field (germination in 2018 was delayed by storm Emma), to capture the more persistent component of the soil seed bank; and in November 2018 (soon after dispersal for most species), to capture the more transient component of the seed bank associated with the most recent seed spread. At each site, five comparable invaded and non-invaded plots (10 in total) were identified and, within each plot, five soil cores were extracted. Soil samples were collected from two depths (0–5 cm and 5–10 cm), resulting in 100 samples per site (600 soil samples per species). The seedling emergence approach (*sensu* Thompson *et al*., 1997) was used to assess soil seed bank density (no. of seedlings per m$^2$), as only the viable component of the seed bank was of interest. Soil samples were processed by removing any vegetative reproductive organs present before potting them into 7 × 7 cm square pots, using a 2:1 (v/v) mix of compost (John Innes no. 2, UK) and sand. Pots were then placed over trays to retain excess water and transferred to semi-controlled environmental conditions in glasshouses at the Rosemount Research Unit, University College Dublin (UCD), in May 2018 (persistent seed bank samples) and in February 2019 (transient seed bank samples, after 3 months of dark and cold storage). Pots were watered biweekly to field capacity. Once a month, samples were stirred to improve aeration and to encourage germination, and their positions were randomly changed to account for environmental variability inside the greenhouse. Seedling emergence was recorded weekly for 12 months, following the procedure described by Gioria and Osborne (2009a). The relationship between the above-ground vegetation and the soil seed bank was also assessed.

To characterise the long-term impact of the study invaders on native plant communities, multiple analytical approaches were used. These included characterisation of changes at the functional level (i.e. whether or not some functional groups had disappeared as a consequence of plant invasions) as well as the susceptibility of native plant communities to plant invasions, by assessing functional and phylogenetic differences between invaded and non-invaded communities. These analyses included testing for changes in diversity associated with invasions by *G. tinctoria* and *I. glandulifera* in the standing vegetation and the soil seed bank through the use of different diversity or dominance indices (species richness, Shannon diversity index, Simpson's diversity index and Pielou's evenness index), assessments of changes in species composition (non-multidimensional scaling, PERMANOVA, similarity percentage analyses, multivariate beta diversity, Procrustes analysis), changes in functional composition (functional traits analysis) and assessments of phylogenetic distances between invaded and non-invaded communities. The phylogenetic tree for the angiosperm taxa in our dataset was reconstructed using the R package V.PhyloMaker (Jin and Qian, 2019).

### *2.2.2 Germination trials*

Germination tests were conducted on seeds of *G. tinctoria* that assessed the effect of environmental conditions on germination. The results from these tests could be important in understanding the factors that contribute to the success of *G. tinctoria* and identifying targets for controlling the establishment of this invader. Germination tests were carried out in growth cabinets in the Earth Institute, UCD, starting in July 2019. Seeds of *G. tinctoria* collected from Achill Island sites were used after treating them with a light bleach solution (10%) to prevent mould formation and to mechanically remove their seed coat. To do that, *G. tinctoria* seeds were placed in a conical flask, with just enough sterile deionised water to cover them, and the water stirred using a magnetic stirrer until most of the seed coats had been removed. Then, 10 replicates of 20 seeds were placed on filter paper within sealed Petri dishes under the following light treatments: (1) full light (210 $\mu mol\,m^{-2}\,s^{-1}$), (2) shade (60% light reduction, 84 $m^{-2}\,s^{-1}$), (3) dark and (4) red filter. The different light levels were tested to investigate the effect of shade on *G. tinctoria* germination rates. We also included a red light treatment, as exposure of seeds to red light has been reported to stimulate germination in some species, such as reed canary grass (*Phalaris arundinacea*) (Junttila *et al.*, 1978) and the common bent (*Agrostis capillaris*) (Williams, 1983). Each light treatment was also divided into two temperature treatments, simulating summer and spring day/night temperatures (20°C and 17°C, 15°C and 12°C, respectively). For the summer simulation treatment the germination rates of *G. tinctoria* were compared with those of the co-dominant native grass, *Agrostis stolonifera*, under (1) full light, (2) 60% full light and (3) dark. Seed germination was monitored for 35 days.

## 2.3 Work Package 3

In this WP, individual-, community- and ecosystem-level investigations were combined to improve our understanding of the cascading effects that the increasing dominance of invasive species may have on soil biochemical cycles and how this could ultimately affect ecosystem functioning. Such an approach is crucial because of the importance of feedback between plants, soils and the decomposer organisms. By measuring a series of plant and soil parameters, potential linkages between the invaded plant community and (1) soil nutrient cycling (e.g. available nitrogen and phosphorus in soils), (2) soil carbon and nitrogen (e.g. carbon concentration but also soil carbon stocks) and (3) soil microbial biomass and activity (i.e. extracellular enzyme activity) were examined.

To assess the impact of *G. tinctoria* and *I. glandulifera* on biogeochemical cycles, the soil legacy impact of plant invasions was examined, at the same sites where the soil seed banks had been assessed (WP2). Samples were collected in June–September 2017, March 2018 and August–October 2018 at the three sites dominated by the invasive species and three adjacent sites associated with a native plant community, as identified in WP2, at both locations (Achill Island, Co. Mayo, and River Lagan, Belfast). Five quadrats were randomly established in each site, and soil samples were collected to examine the impacts of these invaders (five soil samples, each made of four composite samples, within each invaded and non-invaded plot at the study sites), to characterise multiple soil physical, chemical and biological properties. This study design allowed a comparison between invaded and non-invaded communities (Figure 2.3).

Between July 2017 and December 2018, a number of field surveys and soil samplings were conducted to gather information on multiple soil parameters,

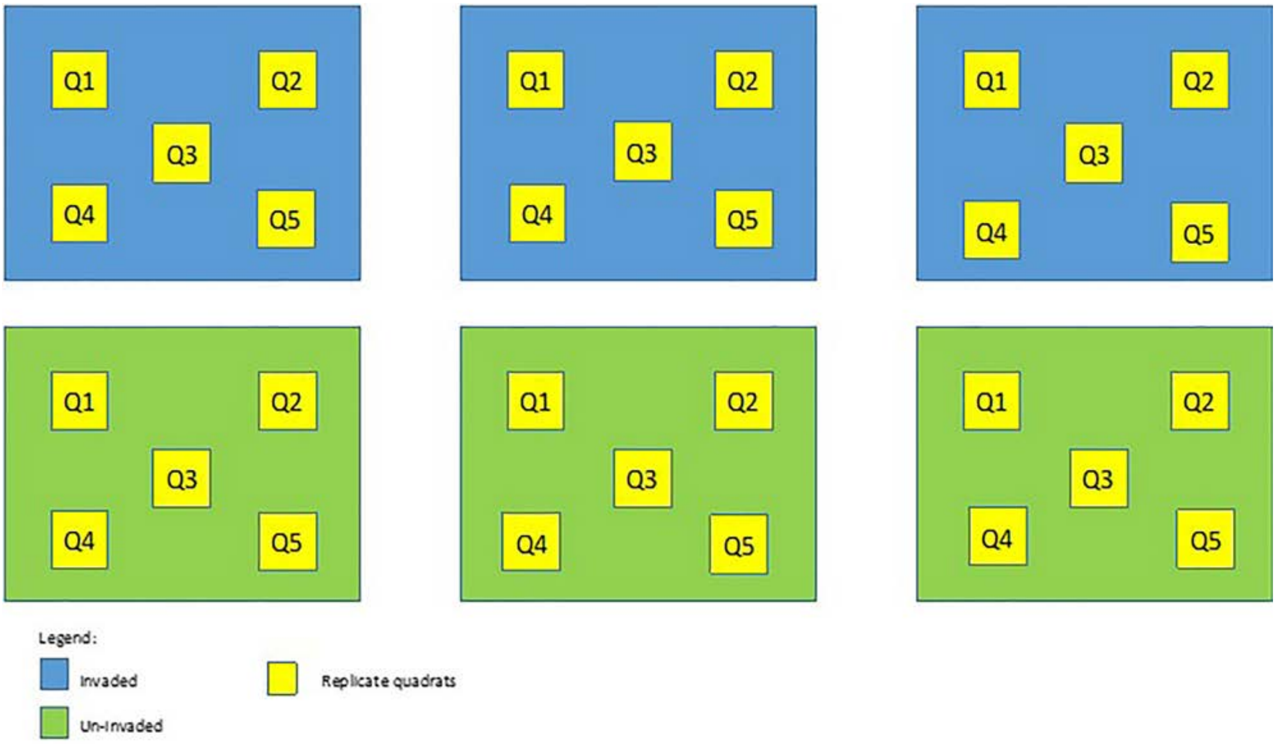

**Figure 2.3. Schematic showing the experimental design used in WP3. Blue boxes represent the invaded plant community (either *G. tinctoria* or *I. glandulifera*) while green boxes represent the native plant community.**

including (1) soil nitrogen and phosphorus availability, (2) total soil carbon and nitrogen content, (3) soil bulk density, (4) soil pH and (5) soil microbial biomass and activity (Table 2.2). Available phosphorus ($PO_4^{3-}$, i.e. Olsen P) in soils was measured by extraction with sodium bicarbonate ($NaHCO_3$). Total carbon and nitrogen were analysed by combustion and gas chromatography. The levels of ammonium ($NH_4^+$-N) and nitrate ($NO_3^-$-N) in soils were measured in soil solutions using a Bran+Luebbe AA3 auto-analyser. The activity of a carbon-acquiring enzyme (i.e. β-1,4-glucosidase) and soil microbial biomass were also measured. The beta-glucosidase enzyme is required for the hydrolysis of cellulose and other beta-linked glucans and could give indications of carbon availability to microbes in soils from invaded and uninvaded plant communities. Similarly, changes in microbial biomass (measured using the fumigation–extraction method) in soils could help in interpreting soil ecosystem response to plant invasion and provide information on changes in resource supply and thus on competitive interactions between invasive alien plants and native species. Analysis of variance (ANOVA) was performed to test for the effects of plant community "invasion status" (invaded vs native community) and soil depth on multiple soil variables.

**Table 2.2. Soil parameters measured and their meaning from a soil ecosystem functioning perspective**

| Soil parameter | Meaning |
|---|---|
| Soil $NO_3^-$, $NH_4$ | Availability of inorganic nitrogen (nitrogen cycling) |
| Soil $PO_4$ (Olsen P) | Availability of inorganic phosphorus (phosphorus cycling) |
| Total soil C | Soil carbon content (%) (soil carbon storage) |
| Total soil N | Soil nitrogen content (%) (soil nitrogen storage) |
| β-1,4-Glucosidase activity | Microbial activity (carbon-acquiring enzyme) |
| Soil pH | Key soil chemical property |
| Soil bulk density | Soil mass per unit volume |

## 2.4 Work Package 4

The original aim of WP4 was to examine the relative role of plant competition and phenological differences between the two selected invaders and their main competitors during the initial stages of invasion. Thus, the establishment of two separate experiments was considered necessary to generate results that could be further used to develop management and restoration measures. Detection of priority effects, i.e. the early emergence of invasive species in the field, could be incorporated into the implementation of any control measures. This should be designed to suppress the growth of the invasive species, which would otherwise germinate earlier than native species, by sowing mixtures of native species that emerge earlier than or at the same time as the invasive species. Furthermore, combining the results of the competition and priority experiments would enable us to understand if priority effects or competition for resources between native grasses and *G. tinctoria* are crucial for the establishment of the invasion and indicate possible ways of suppressing the introduced species.

For these experiments, seeds of *G. tinctoria* were collected in August, October and December 2018 from Achill Island sites, and seeds of *I. glandulifera* were collected from the River Lagan site in October 2018 (see WP2 for site descriptions). Seeds of both species were subjected to 2 months of cold, dry storage. Following this, the seed coat was mechanically removed from some *G. tinctoria* seeds (see section 2.2.2 for method description), while some *I. glandulifera* seeds were subjected to 7 days of cold, wet stratification (Perglová *et al*., 2009). For both species, germination trials were carried out under growth chamber and greenhouse conditions, using various germination substrates (filter paper in Petri dishes, vermiculite, compost, sand in pots). Germination of *G. tinctoria* was unpredictable and varied greatly depending on several factors, such as the germination substrate, whether the study was conducted in growth chambers or in a greenhouse, and whether the seed coat had been removed or not, with an average total germination of around 50%. No germination of *I. glandulifera* occurred in any of the treatments, substrates or conditions used, so *I. glandulifera* could not be used in the competition experiments.

The study used the native grass *A. stolonifera* as the main competitor of *G. tinctoria* rather than a species mix, as had been originally planned, as this species is the dominant grass in coastal areas on Achill Island invaded by *G. tinctoria* and similar grassland communities elsewhere, and makes up the largest proportion of the total biomass of the community. Furthermore, previous work has shown that the dominant species identity has a strong effect on community invasibility (Emery and Gross, 2007), thereby justifying an approach that compares competitive interactions between *G. tinctoria* and *A. stolonifera*.

### *2.4.1 Competition experiment*

Because of the difficulties caused by the unpredictable timing and rates of germination of *G. tinctoria* seeds, it was decided to work with *G. tinctoria* seedlings from seeds germinated under fully controlled environmental conditions in growth chambers. Similarly, unpredictability in the timing of *G. tinctoria* germination made it impossible to replicate the priority conditions observed in the field. For this reason, the study used *A. stolonifera* above-ground biomass as a proxy for the competitive pressure that this species probably exerts on *G. tinctoria* seedlings in the field. Thus, in August 2019, three *G. tinctoria* seedlings of the same age and size (2-week-old seedlings) were planted 2 weeks after sowing 0.05 g of *A. stolonifera* seeds per half-tray, making 40 half-trays in total. To simulate different levels of competition, four biomass removal treatments were created through repetitive clipping of the shoots of *A. stolonifera* to maintain the height difference between the two species at four pre-set levels. The treatments included (1) no competition, where the three *G. tinctoria* seedlings were grown on their own; (2) low competition, with a 1 cm height difference; (3) intermediate competition, with a 6 cm height difference; and (4) high competition, with a 12 cm height difference between the two species (Figure 2.4). Additionally, to assess the effect of resource availability on competition, nutrients were added weekly to half of the replicates as an NPK (nitrogen, phosphorus, potassium) solution (Bratan 18-18-18 + 3 MgO + trace elements, at half-strength).

The initial mortality of *G. tinctoria* seedlings, between August and November 2019, was very high, at more than 50%. During this 4-month period, mortality was similarly high for all experimental treatments, and PERMANOVA revealed no significant difference in mortality depending on competition or nutrient levels ($p = 0.9954$ and $p = 0.2457$, respectively). To continue the experiment, it was decided to replace dead seedlings with seedlings of similar age and size growing in a separate seedling pool under the same conditions. In November 2019, we stopped replacing the dead seedlings and the probability of survival was

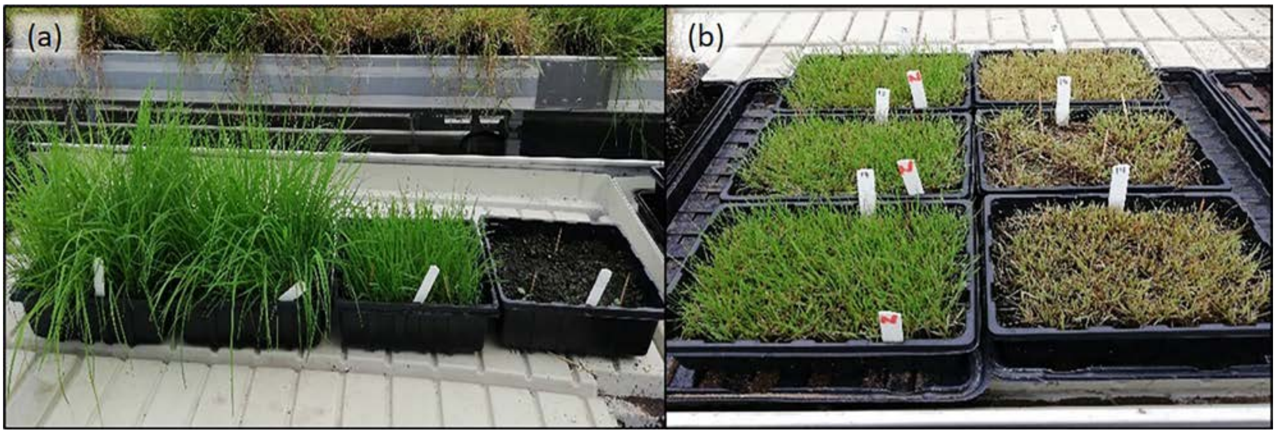

**Figure 2.4. Experimental set-up for the competition experiments with three *G. tinctoria* seedlings and 0.05 g of *A. stolonifera* per half-tray. (a) Four treatments, from left to right: high, intermediate, low and no competition. (b) Six plots assigned to the low-competition treatment; the three on the left-hand side of the picture were fertilised and the three on the right-hand side were not fertilised.**

then defined as the number of seedlings that survived the last 2 months of the experiment.

The leaf area (LA) of *G. tinctoria* was estimated using three different equations, depending on their linear dimensions (length, L; breadth, B): for leaves with L or B smaller than 2 mm, the equation for the area of an ellipse was used (equation 2.1); for leaves with L or B between 2 and 48 mm, the study used the equation for the area of a hexagon (equation 2.2); and for leaves with L or B higher than 48 mm, we used equation 2.3 (see Mantoani, 2019, 2020):

$$LA = \pi \times (B/2) \times (L/2) \tag{2.1}$$

$$LA = 3 \times (B/2) \times (B/2) \tag{2.2}$$

$$LA = ((0.7276 \times L \times B) + 19.538) \times 0.5 \tag{2.3}$$

At the end of February 2020, the experiment was harvested, and the study compared the performance of *G. tinctoria* across different competition and fertility treatments, in terms of the probability of survival, plant biomass and its components (leaves, petioles, rhizome and roots), leaf number and leaf area. The study also assessed *A. stolonifera* above-ground biomass. The results were analysed using PERMANOVA. The dried biomass was ground and analysed for stable isotopes ($\delta^{15}N_{AIR}$ and $\delta^{13}C_{V\text{-}PDB}$), as well as the concentrations of nitrogen, carbon, macronutrients (potassium, phosphorus, magnesium, calcium and sulfur) and micronutrients (sodium, iron, manganese, copper and zinc), to assess how competition/fertilisation influenced nutrient uptake and utilisation. Carbon and nitrogen isotopes were determined using mass spectrometry, and elemental analysis was determined using inductively coupled plasma optical emission spectroscopy (ICP-OES). In the case of *G. tinctoria*, the available biomass was sufficient only for the analysis of the no- and low-competition treatments. Additionally, the researchers assessed the degree of infection of *G. tinctoria* rhizomes by the symbiotic cyanobacterium *Nostoc punctiforme* by cutting a cross-section of the rhizomes and taking an image of their maximum diameter using the SketchAndCalc mobile phone application to calculate the percentage area infected.

### 2.4.2 Early germination and leaf-out/priority effects

The study assessed priority effects between *G. tinctoria* and the native, co-dominant grass, *A. stolonifera,* to improve our understanding of potential mechanisms underlying successful invasions in coastal grasslands in Ireland. For this experiment, seeds collected from Achill Island sites (WP2), were sown in half-trays in glasshouses at the Rosemount Experimental Station, UCD, in May 2019, under three conditions:

1. Seeds of *G. tinctoria* were sown before seeds of the native, co-dominant grass, *A. stolonifera* (early treatment).
2. Seeds of the invasive and the native species were sown simultaneously (intermediate treatment).

3. Seeds of the native species were sown before those of the invasive species (late treatment).

To account for differences in the timing of seed germination between the two competing species, we tested the average seed viability and germination time in growth chamber trials. Differences in germination characteristics derived from these trials were incorporated into the experimental design. Unfortunately, under glasshouse conditions, germination did not occur as expected. Germination of *G. tinctoria* was not only very low (less than 1%) but did not begin for over 2 months. By that time, *A. stolonifera* had fully germinated and grown significantly. Given the failure of the timing manipulations, it was decided to abandon this experimental design and to resort to the experiment described below.

To avoid the unpredictability of *G. tinctoria* germination, we used seedlings that had been germinated previously in growth chambers. The effects of phenological differences were examined in half-trays at the greenhouse facility at the UCD Rosemount Environmental Station, starting in July 2019. Three priority conditions were recreated, with seedlings of *G. tinctoria* planted 16, 23 and 33 days after the sowing of the native, co-dominant grass, *A. stolonifera.* The duration of the priority treatments was based on field observations and considered the average germination time of *A. stolonifera* seeds. As in the competition experiment, the initial mortality of *G. tinctoria* seedlings was around 50% and dead seedlings were replaced (up to October 2019) with seedlings of similar age growing in pots under comparable greenhouse conditions.

The number of leaves of *G. tinctoria* seedlings was measured in November 2019 and in March, April, June, July and August 2020. Their leaf area was calculated based on measurements of linear dimensions, as described in section 2.4.1. In August 2020, we harvested the experiment and compared the performance of *G. tinctoria* across different competition and fertility treatments, in terms of the probability of survival, plant biomass (separated into leaves, petioles, rhizome and roots), leaf number and leaf area. Comparisons were made with *A. stolonifera* biomass production. The results were analysed using PERMANOVA. Additionally, the study assessed the degree of infection of *G. tinctoria* rhizomes by the symbiotic cyanobacterium *N. punctiforme*, as described in section 2.4.1.

## 2.5 Work Package 5

This WP was aimed at developing control and eradication measures that are effective and sustainable over time. For this purpose, we combined the results of WPs 1–4, so that the management protocols would be based on information on:

1. the size and type of seed bank formed by invasive species;
2. the impact of plant invasions at the community and ecosystem levels;
3. the competitive ability of the invasive species compared with that of the native species (thus contributing to information on the resilience of native communities to plant invasions);
4. potential priority effects that may favour the establishment of early-emerging invasive species;
5. previous knowledge of management measures used elsewhere under conditions similar to those experienced in Ireland, with a review of their effectiveness.

The technical note described in the results from WP1 was used to produce a final review of the most effective measures that can be used to manage rhizome-forming alien species and species forming persistent seed banks.

Based on the results of WP2 and WP3, the study also provided recommendations on how to develop restoration measures that will account for the varying characteristics of native and invasive species, including the nature of their seed banks and competitive ability, as well as the impact of invasive species, and taking account of priority effects. Depending on the characteristics and long-term impact of invasive plants, potential restoration measures were developed that not only suppress and prevent the re-establishment of invasive species but would also be effective in returning the invaded areas to pre-invasion conditions where this is possible. The study evaluated the potential for secondary invasions by other alien species, and whether primary invasions are a cause or an effect of environmental degradation.

# 3 Results and Discussion

## 3.1 Work Package 1

Results from WP1 were published in a paper entitled "A review of the impacts of major terrestrial invasive alien plants in Ireland" (Gioria *et al*., 2018b). This was a comprehensive review of current knowledge of the ecological impacts of terrestrial invasive alien plants, including changes at the community and ecosystem levels, the extent of hybridisation with native or other alien species, and likely secondary invasions by alien plants. The review also includes the projected economic costs of terrestrial plant invasions, as estimated in relevant studies, and information on the impacts of invasive alien plants in protected areas. This paper ended with a discussion of the potential long-term impacts of plant invasions on Ireland's native biodiversity and ecosystems, considering the increasing spread of invasive plant species associated with socio-economic and environmental changes.

Reviewing knowledge of the impacts of terrestrial plant invasions on native communities and ecosystems showed that there is a lack of quantitative information on this topic. Most studies or reports have observed decreases in plant diversity associated with plant invasions, but there is little information on the identity of the species that are most affected by plant invasions. There is also little information on potential local or national extinctions, on changes in the functional and genetic diversity of invaded communities, and on changes at different trophic levels, other than primary producers (plants). There is a lack of information on the impacts of invasive alien plants at the ecosystem level, such as changes in biogeochemical cycles. We know from the literature that these changes could play a critical role in promoting the persistence and spread of primary invaders and facilitating secondary invasions by other alien plants. Knowledge of the recovery potential of native species after the removal of an invader is also lacking. These changes are likely to interact with climatic and other environmental changes, by affecting the probability of successful establishment and spread of new alien species and current invaders, and influence the capacity of native species to respond to the new conditions introduced by the invasive species.

Further results were published in a second paper, entitled "Managing invasions by terrestrial alien plants in Ireland: challenges and opportunities" (Gioria *et al*., 2019). The authors reviewed strategies that have been put in place to manage terrestrial invasive alien plants in Ireland, and the paper describes the principles that should guide the creation of a biosecurity infrastructure that is effective at preventing the introduction of propagules of invasive or potentially invasive alien plants, including the development of an early identification and rapid intervention system. The principles that should guide the development of cost-effective species- and habitat-specific control (or eradication) and restoration programmes are also described. The paper concludes by discussing the challenges and opportunities associated with the management of terrestrial plant invasions in Ireland.

A technical note on the management of *G. tinctoria* for the International Union for Conservation of Nature (IUCN) was completed in July 2017 (Gioria and Osborne, 2017). This 16-page report described the most cost-effective measures that have been used to control this species where it is invasive globally. It also described the novel approaches that could be used to (1) prevent its introduction into EU Member States where is not currently present and (2) detect new individuals or populations before they reach sexual maturity and potential measures that can be used for the rapid eradication of these individuals/populations. It furthermore describes novel approaches that are effective at controlling this species, including measures that account for its capacity to accumulate a large, persistent soil seed bank and an ability for extensive vegetative propagation. This report was published in September 2017 on the CIRCABC (Communication and Information Resource Centre for Administrations, Businesses and Citizens) website (https://circabc.europa.eu), which is run by the European Commission. This technical note was used to discuss the need for improved measures for the management and the disposal of current high-impact invasive alien plants in Ireland.

**Table 3.1. List of species found in the standing vegetation on Achill Island and in Lagan Valley Regional Park invaded (I) and uninvaded (U) sites**

| | Achill Island | | | | | | Lagan Valley Regional Park | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Site 1 | | Site 2 | | Site 3 | | Site 1 | | Site 2 | | Site 3 | |
| Species | I | U | I | U | I | U | I | U | I | U | I | U |
| *Achillea millefolium* | | X | | X | | | | | | | | |
| *Agrostis capillaris* | | | X | | | | | | | X | | |
| *Agrostis stolonifera* | | | X | X | X | X | X | X | X | X | X | X |
| *Alopecurus geniculatus* | | X | | | | | | | | | | |
| *Anthriscus sylvestris* | | X | X | X | X | X | | | | X | | X |
| *Arctium minus* | | X | | | | | | | | | | |
| *Atriplex prostrata* | | X | | | | | | | | | | |
| *Bellis perennis* | | X | | | | | | | | | | |
| *Cardamine flexuosa* | | | | X | | | | | | | | |
| *Carex otrubae* | | | | X | | | | | | | | |
| *Cerastium fontanum* | | X | | | | | | | | | | |
| *Cirsium arvense* | | X | | X | | | | | | X | | |
| *Cirsium palustre* | | X | | X | | | | | | | | |
| *Cirsium vulgare* | | X | | X | | | | | | | | |
| *Crocosmia × crocosmiiflora* | | | | | X | X | | | | | | |
| *Dactylorhiza incarnata* | | | | X | | | | | | | | |
| *Epilobium hirsutum* | | X | | X | | X | | | | | | |
| *Epilobium palustre* | | X | | X | | | | X | | X | | X |
| *Equisetum arvense* | | | | X | | X | | | | | | |
| *Equisetum fluviatile* | | | X | X | | | | | | | | |
| *Eryngium maritimum* | | X | | | | | | | | | | |
| *Fallopia japonica* | | X | | | | X | | | | | | |
| *Festuca rubra* | X | X | | | | X | | | | | | |
| *Filipendula ulmaria* | | | | | | | | | | | | X |
| *Galium aparine* | X | X | X | X | X | | | | X | X | X | X |
| *Gunnera tinctoria* | X | | X | | X | | | | | | | |
| *Hedera hibernica* | | | | | | | | | | | X | X |
| *Heracleum mantegazzianum* | | | | | | | | X | | X | | |
| *Heracleum sphondylium* | | | | | | | | X | | X | | X |
| *Holcus lanatus* | | X | | X | | X | | | | | | |
| *Impatiens glandulifera* | | | | | | | X | | X | | X | |
| *Juncus bufonius* | | X | | | | X | | | | | | |
| *Juncus effusus* | | X | | X | | X | | | | | | |
| *Lolium perenne* | | | | | | X | | | | | | |
| *Lotus corniculatus* | | X | | | | | | | | | | |
| *Lythrum salicaria* | | X | | X | | X | | | | | | |
| *Plantago major* | | | | | | | X | | | | | |
| *Polygonum persicaria* | | X | | | | X | | | | | | |
| *Potentilla anserina* | | X | | X | | X | | | | | | |
| *Prunella vulgaris* | | X | | | | X | | | | | | |
| *Ranunculus repens* | | X | | X | X | X | X | X | X | X | X | X |
| *Rosa canina* | | X | | | | | | | | | | |
| *Rubus fruticosus* | | X | | | | X | | | X | | | |
| *Rumex conglomeratus* | | X | | | | | | | | | | |

**Table 3.1. Continued**

| | Achill Island | | | | | | Lagan Valley Regional Park | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Site 1 | | Site 2 | | Site 3 | | Site 1 | | Site 2 | | Site 3 | |
| **Species** | I | U | I | U | I | U | I | U | I | U | I | U |
| *Rumex crispus* | | X | | X | | | | | | | | |
| *Rumex obtusifolius* | | | | | | | X | X | X | X | | X |
| *Rumex sanguineus* | | X | | | | X | | | | | | |
| *Senecio aquaticus* | | X | | | | | | | | | | |
| *Silene vulgaris* | | | | | | X | | | | | | |
| *Sonchus arvensis* | | X | | | | | | | | | | |
| *Stachys palustris* | X | X | | | | X | | | | | | |
| *Taraxacum officinale* | X | X | | | | | X | | | | X | X |
| *Trifolium hybridum* | | | | X | | | | | | | | |
| *Trifolium pratense* | | X | | X | | | | | | | | |
| *Trifolium repens* | | X | | X | | X | | | | | | |
| *Urtica dioica* | X | X | X | X | X | X | X | X | X | X | X | X |

## 3.2 Work Package 2

### *3.2.1 Above- and below-ground community impacts*

The results of the above-ground community assessments indicated that *G. tinctoria* and *I. glandulifera* differed substantially in their impact on the standing vegetation. *G. tinctoria* invades species-rich (22–37 species; for a list of species see Table 3.1) coastal grassland communities and had a strong impact on the standing vegetation, resulting in an average reduction in plant species richness of 74% (Figure 3.1). In doing this, it converted

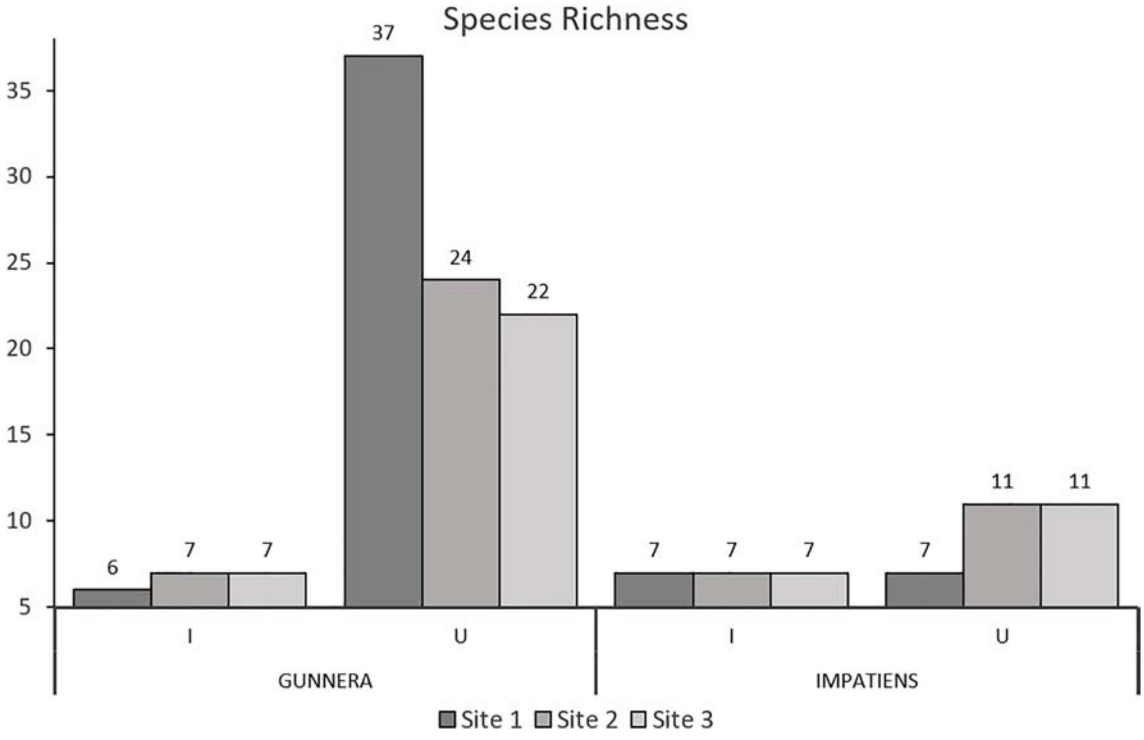

**Figure 3.1. Total number of species found in the standing vegetation in invaded and comparable uninvaded plots at three sites (S1–S3) invaded by *G. tinctoria* and *I. glandulifera*. I, invaded; U, uninvaded.**

these communities into species-poor communities dominated by the invader itself and by weedy species, such as *U. dioica*, which was consistently found associated with *G. tinctoria* in invaded plots. In contrast, *I. glandulifera* invaded species-poor riparian plant communities (7–11 species), with an average reduction in plant species richness of 21%. However, the impact of *I. glandulifera* on the species richness and the dominance or evenness index was not significant. Interestingly, the invaded plant communities had a similar species richness despite differences in the identity of the invaders, the characteristics of the invaded habitats (coastal grasslands vs riparian habitats) and the initial (pre-invasion) species richness. Multivariate analyses of abundance data (cover) confirmed that the impact on the standing vegetation was highly significant for *G. tinctoria*. *I. glandulifera* also significantly altered the composition of the resident communities, but mainly by replacing a community dominated by *U. dioica* with one dominated by *I. glandulifera* (Figure 3.2). Uninvaded communities associated with *I. glandulifera* supported an invasive population of giant hogweed, *H. mantegazzianum*, which is listed as a species of Union concern and is known for its detrimental impacts on native communities and human health.

Assessment of changes in the seed bank showed that both invaders had a significant impact on the persistent and the transient seed bank species composition, which reflected changes in the standing vegetation (Figure 3.3; for a list of species see Tables 3.2 and 3.3). However, seed banks supported significantly more species than the standing vegetation, with an average of 27 and 37 species, respectively, in invaded and uninvaded seed banks associated with *G. tinctoria* invasions, while on average 29 and 32 species, respectively, were found in invaded and uninvaded seed banks associated with *I. glandulifera* (Figure 3.4). These results supported evidence of the importance of soil seed banks as a strategy to buffer against unfavourable environmental conditions. For *G. tinctoria*, invaded seed banks supported a lower number of species and a higher proportion of seeds of ruderal than in uninvaded communities. Interestingly, in site 1, seed banks invaded by *I. glandulifera* supported a greater number of species than the adjacent uninvaded areas. However, most of these species were pioneer (ruderal) species, which typically form larger and more persistent seed banks than later successional species. Perhaps the presence of invasive species has increased the proportion of ruderal species found in

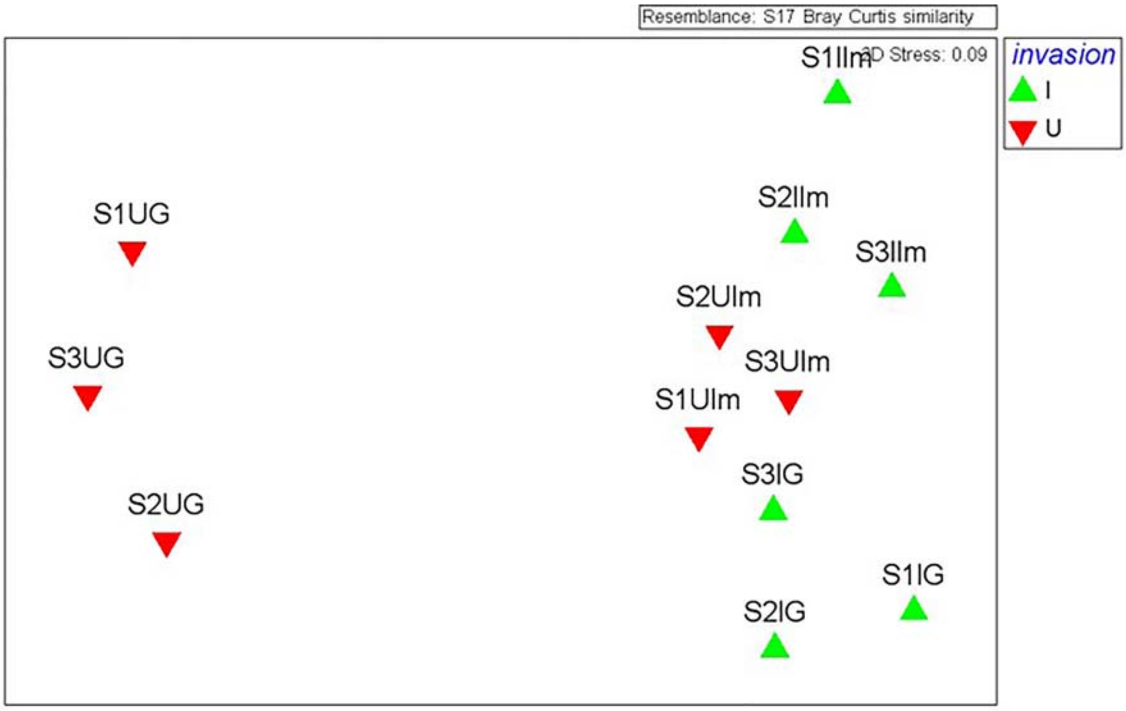

**Figure 3.2. Non-multidimensional scaling plot showing differences in community composition in invaded (I) and comparable uninvaded (U) plots (data are averaged for each plot) at three sites (S1–S3); *G. tinctoria* (G) and *I. glandulifera* (Im).**

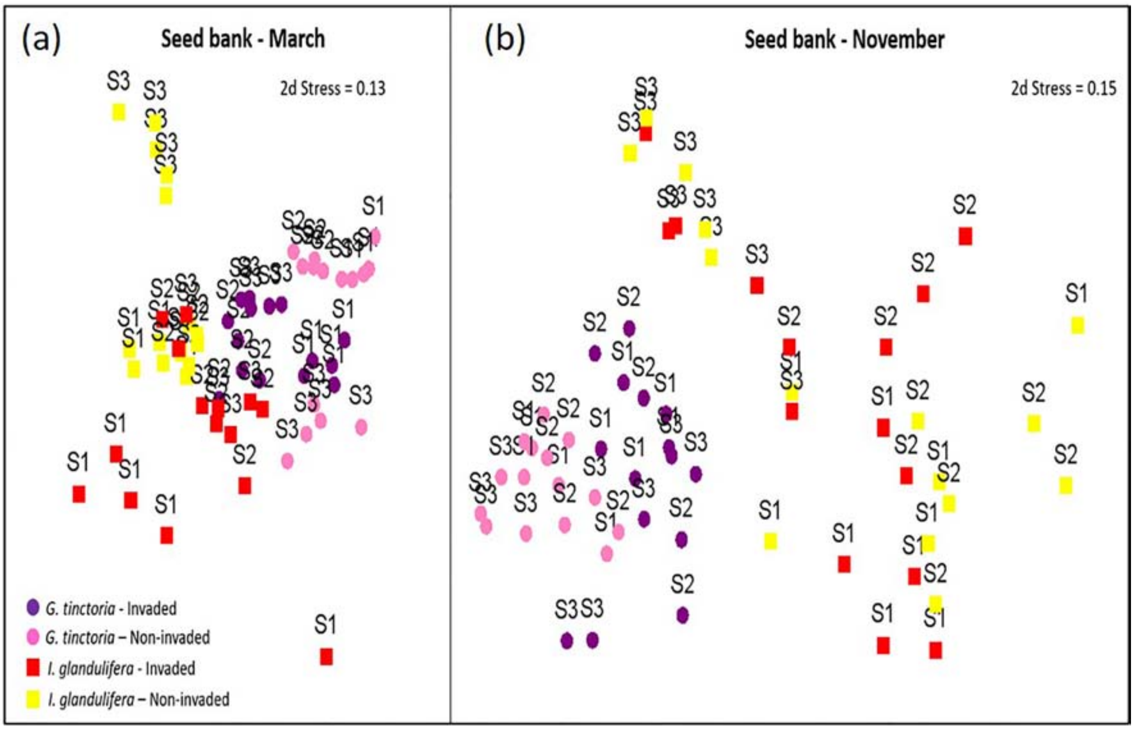

**Figure 3.3. Non-multidimensional scaling plot showing differences in the composition of the soil seed bank collected in (a) March (persistent seed bank) and (b) November (transient seed bank) in invaded and comparable uninvaded plots (data are averaged for each plot) at the three sites invaded by *G. tinctoria* and *I. glandulifera*.**

the area, resulting in more species in the seed bank, supporting global evidence that disturbed or stressed plant communities tend to be composed of a greater proportion of persistent seeds than relatively pristine plant communities (Kiss *et al*., 2017).

The findings of this study will have important implications for the management of these invaders. Firstly, they show that these invaders differ substantially in the impact they exert on plant community types, although both are listed in the EU IAS Regulation. *G. tinctoria* promotes longer term changes in both the above-ground and below-ground flora in species-rich grassland communities. The results show that *I. glandulifera* tends to become invasive in species-poor, degraded communities, and that the overall impact on the standing vegetation is minimal. Moreover, *I. glandulifera* displaced giant hogweed, *H. mantegazzianum*, which was present in uninvaded, but not in invaded, areas, both in the standing vegetation and in the seed bank. Given the risk to health posed by giant hogweed, these findings would suggest a beneficial effect of *I. glandulifera* on the invaded communities. However, its removal would probably promote colonisation by *U. dioica* and other alien or weedy plants. There were also contrasting effects on the seed banks, with *G. tinctoria* forming long-term persistent seed banks, while *I. glandulifera* formed only transient ones, indicating that the management of the seed bank of the former would require extensive efforts and multiple measures repeated over several years, which might not be feasible. In contrast, re-establishment of *I. glandulifera* could be prevented in a relatively short period of time, by preventing the dispersal of new seeds.

### *3.2.2 Germination trials*

The results of the germination trials with different light and temperature treatments revealed a significant association between *G. tinctoria* germination and temperature ($p = 0.0001$), with 62% germinating in summer and 45% in spring (Figure 3.5). Furthermore, total germination was significantly inversely associated with irradiance ($p = 0.0286$), but only in comparisons between full light (47%) and darkness (57%) (Figure 3.6). By contrast, total germination in the shade treatment or under red light did not differ significantly from that observed with the other

**Table 3.2. List of species found in the transient seed bank on Achill Island and in Lagan Valley Regional Park invaded (I) and uninvaded (U) sites**

| | Achill Island | | | | | | Lagan Valley Regional Park | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Site 1 | | Site 2 | | Site 3 | | Site 1 | | Site 2 | | Site 3 | |
| Species | I | U | I | U | I | U | I | U | I | U | I | U |
| *Agrostis canina* | X | X | X | X | X | X | | | | | X | X |
| *Agrostis capillaris* | | | | | | X | | | | | X | X |
| *Agrostis pratensis* | | | | | | | X | X | | | | |
| *Agrostis stolonifera* | | X | | | | | | | | | | |
| *Alopecurus geniculatus* | X | X | X | X | X | X | X | X | X | X | X | X |
| *Alopecurus pratensis* | | X | | | | | | | | | | |
| *Anthriscus sylverstris* | | | X | X | | | X | | | | X | |
| *Apium nodiflorum* | | | | | X | X | | | | | X | X |
| *Callitriche stagnalis* | | | | X | X | X | | | | | X | X |
| *Cardamine flexuosa* | X | X | | X | | | | | | X | | X |
| *Cardamine hirsuta* | | | | | | X | | | X | X | | |
| *Carex otrubae* | | X | | X | X | | X | X | | | | |
| *Cerastium fontanum* | | X | | X | | X | | | X | X | X | X |
| *Chenopodium album* | | | | | X | | | | | | | |
| *Cirsium arvense* | | | | X | X | X | | | X | X | | |
| *Dactylis glomerata* | | | | | | X | | | | | X | X |
| *Danthonia decumbens* | | | | | X | X | | X | | | X | X |
| *Epilobium cilliatum* | | | | | | | | | | X | X | X |
| *Epilobium hirsutum* | | X | X | X | X | X | X | X | | | | |
| *Epilobium montanum* | | | X | | | X | | | X | X | | |
| *Epilobium obscurum* | X | | X | X | X | X | | | X | X | | |
| *Equisetum arvense* | | | | X | X | | | | | | | |
| *Festuca ovina* | X | | | | | | | | | | | |
| *Galium aparine* | | | | X | | | | | | | X | X |
| *Galium palustre* | | | | X | | X | | | | | X | X |
| *Geranium molle* | | | | | | | | X | | | | |
| *Geranium robertianum* | | | | | | | X | | | | | |
| *Geum urbanum* | | | | | | | X | X | X | X | X | X |
| *Glyceria fluitans* | | X | | | | X | | | | | | |
| *Gnaphalium uliginosum* | | | | | | | X | | | | | |
| *Gunnera tinctoria* | X | X | X | X | X | X | | | | | | |
| *Holcus lanatus* | | | | X | X | X | | | | | | X |
| *Impatiens glandulifera* | | | | | | | X | X | X | X | X | X |
| *Isolepis setacea* | | | | | | X | | | X | X | | |
| *Juncus bufonius* | X | X | X | X | X | X | X | X | X | X | X | X |
| *Juncus bulbosus* | | | | | | X | X | | | | | |
| *Juncus effusus* | X | X | X | X | X | X | X | X | X | X | X | X |
| *Lythrum salicaria* | | X | X | X | X | X | | | X | X | | |
| *Medicago lupulina* | | | X | X | | X | | | | | | |
| *Montia fontana* | X | X | X | X | X | X | | | | X | X | X |
| *Phleum pratense* | | X | | | | | | | | | | |
| *Plantago major* | | | | | | | | | | | X | X |
| *Poa annua* | X | X | X | X | X | X | X | X | X | X | X | X |
| *Poa pratensis* | | | | | | | | X | | | | |

**Table 3.2. Continued**

| | Achill Island | | | | | | Lagan Valley Regional Park | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Site 1 | | Site 2 | | Site 3 | | Site 1 | | Site 2 | | Site 3 | |
| **Species** | I | U | I | U | I | U | I | U | I | U | I | U |
| *Poa trivialis* | | | | X | | | | | | | | |
| *Potentilla anserina* | X | X | | | | | | | | | | |
| *Ranunculus acris* | X | X | | | | | | | | | | |
| *Ranunculus bulbosus* | | | X | | | | | | | | | |
| *Ranunculus repens* | X | X | X | X | X | X | X | X | X | X | X | X |
| *Rubus fruticosus* | | | | | | | X | X | | X | | X |
| *Rumex acetosella* | | | | | X | | | | | | | |
| *Rumex crispus* | X | X | | X | X | X | X | X | X | X | X | |
| *Rumex obtusifolius* | | | X | | | X | | | X | X | X | X |
| *Rumex sanguineus* | | | | | | | X | | X | X | | |
| *Sagina apetala* | | X | X | X | X | X | X | | | X | X | X |
| *Sonchus asper* | X | | X | | X | | | | | | | X |
| *Stachys palustris* | | | X | X | | | | | | | | |
| *Stellaria media* | X | X | X | X | X | X | X | X | X | X | X | X |
| *Trifolium pratense* | | X | | | X | | | | | | | |
| *Tussilago farfara* | | | | | | | X | X | | X | X | X |
| *Urtica dioica* | X | | X | X | X | X | X | X | X | X | X | X |
| *Veronica beccabunga* | | | | | | | X | | | X | | |
| *Veronica hederifolia* | | | | | | | | | X | X | | |
| *Vulpia bromoides* | X | X | X | X | X | X | X | X | X | X | X | X |

treatments. No significant interactions between season and irradiance occurred (Table 3.1). When comparing *G. tinctoria* with *A. stolonifera* exposed to spring temperatures under full light, shade and darkness, there was a significant interaction ($p = 0.0016$) between species and irradiance. While both species had a significantly higher total germination in darkness (51% for *G. tinctoria* and 65% for *A. stolonifera*) than under full light (37% and 33%, respectively), germination in shade remained high for *G. tinctoria* (48%) and was significantly different from germination in the light ($p = 0.0469$) (Table 3.4). In contrast, for *A. stolonifera*, germination in the shade was significantly reduced (27%) and was similar to that under full light but significantly different from germination in the dark ($p = 0.0221$). The comparison between the two species demonstrated that, although both species reach their maximum germination in the dark, germination of *G. tinctoria* was less inhibited by irradiance than germination of *A. stolonifera* (Figure 3.7). These results are interesting, as they demonstrate a partial tolerance of *G. tinctoria* germination to light, in contrast to the native grasses. Thus, the germination of seeds of *G. tinctoria* on the soil or on the rhizome surface would be less inhibited by exposure than seeds of *A. stolonifera*.

The results for germination speed were comparable to those for total germination. Specifically, the speed of germination of *G. tinctoria* was significantly increased at higher temperature ($p = 0.0001$) (Figure 3.8) and significantly affected by light, with germination occurring significantly more slowly under full light than under shade and red light (Figure 3.9). No significant interaction between temperature and light was observed (Table 3.5). Comparison of speed of germination by *G. tinctoria* and *A. stolonifera* in spring also revealed a significant interaction between species and light ($p = 0.0001$) (Table 3.5). Germination of both species was significantly faster in the dark than under full light, but germination in the dark was significantly faster than germination in the shade only in the case of *A. stolonifera* (Figure 3.10).

**Table 3.3. List of species found in the persistent seed bank on Achill Island and in Lagan Valley Regional Park invaded (I) and uninvaded (U) sites**

| | Achill Island | | | | | | Lagan Valley Regional Park | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Site 1 | | Site 2 | | Site 3 | | Site 1 | | Site 2 | | Site 3 | |
| Species | I | U | I | U | I | U | I | U | I | U | I | U |
| *Agrostis capillaris* | | | | | | X | | | X | X | | X |
| *Agrostis stolonifera* | | | X | X | X | X | X | X | X | X | X | X |
| *Alopecurus geniculatus* | | X | | X | X | X | | | | | | X |
| *Anthriscus sylvestris* | | | | | X | X | | | | | | |
| *Apium nodiflorum* | | | | | X | X | | | | | | |
| *Cerastium fontanum* | | | | X | | | | | | | | |
| *Chenopodium album* | | X | X | | | X | X | | | | | |
| *Cirsium arvense* | | X | | | | X | | | | | X | X |
| *Epilobium montanum* | X | X | X | X | X | | X | X | X | X | X | |
| *Equisetum arvense* | | | | | | X | | | | | | |
| *Festuca ovina* | X | X | X | X | X | X | X | | X | X | | X |
| *Geranium robertianum* | | | | | | | | X | | | | |
| *Gunnera tinctoria* | X | X | X | X | X | X | | | | | | |
| *Impatiens glandulifera* | | | | | | | X | | X | | | |
| *Lythrum salicaria* | | | | X | | | | | | | | |
| *Medicago lupulina* | | | | X | | X | | | | | | |
| *Persicaria maculosa* | | X | | | | | | | | | | |
| *Poa annua* | X | X | X | X | X | X | | X | | X | X | |
| *Polygonum aviculare* | | X | | X | X | X | | | | X | | X |
| *Potentilla anserina* | | X | | | | | | | | | | |
| *Prunella vulgaris* | | | | | | X | | | | | | |
| *Ranunculus acris* | | | | | | | | | | X | | |
| *Ranunculus repens* | X | X | X | X | X | X | | X | X | X | X | X |
| *Rumex crispus* | | X | | | | | | | | | | |
| *Sagina apetala* | | X | | | | | X | X | | | X | X |
| *Senecio jacobaea* | | | | | | | | | X | | | X |
| *Sonchus oleraceus* | | X | X | X | X | X | X | X | X | X | X | X |
| *Stellaria media* | | | X | X | X | X | | | X | X | | |
| *Trifolium pratense* | | | X | | | | | | | X | | |
| *Tussilago farfara* | | | | | | | | X | | X | | |
| *Urtica dioica* | X | X | X | X | X | X | X | X | X | X | X | X |
| *Veronica hederifolia* | | X | X | | | | | | | | | |
| *Veronica persica* | | | | | X | X | X | | X | | X | X |

## 3.3 Work Package 3

The study found that soil phosphorus availability (Olsen P) measured in January 2018 on Achill Island was not significantly affected by invasion ($p = 0.97$), but that it significantly decreased with soil depth from 50 ± 5 mg P $kg^{-1}$ soil at depths of 0–5 cm to 29 ± 3.1 mg P $kg^{-1}$ soil at depths of 5–20 cm ($p = 0.006$; Figure 3.11a). However, soil phosphorus availability measured at the River Lagan sites was significantly affected by invasion (invaded vs native community; 12.8 ± 1.5 vs 20.3 ± 1.7 mg P $kg^{-1}$ soil) ($p = 0.0009$) and significantly decreased with soil depth ($p < 0.0001$; Figure 3.11b). No significant interactions were found between invasion and soil depth.

The results show that in September 2017 invasion status significantly affected soil $NO_3^-$ and $NH_4$ levels ($p < 0.0001$) on Achill Island. Soil $NO_3^-$ levels were significantly higher in invaded areas

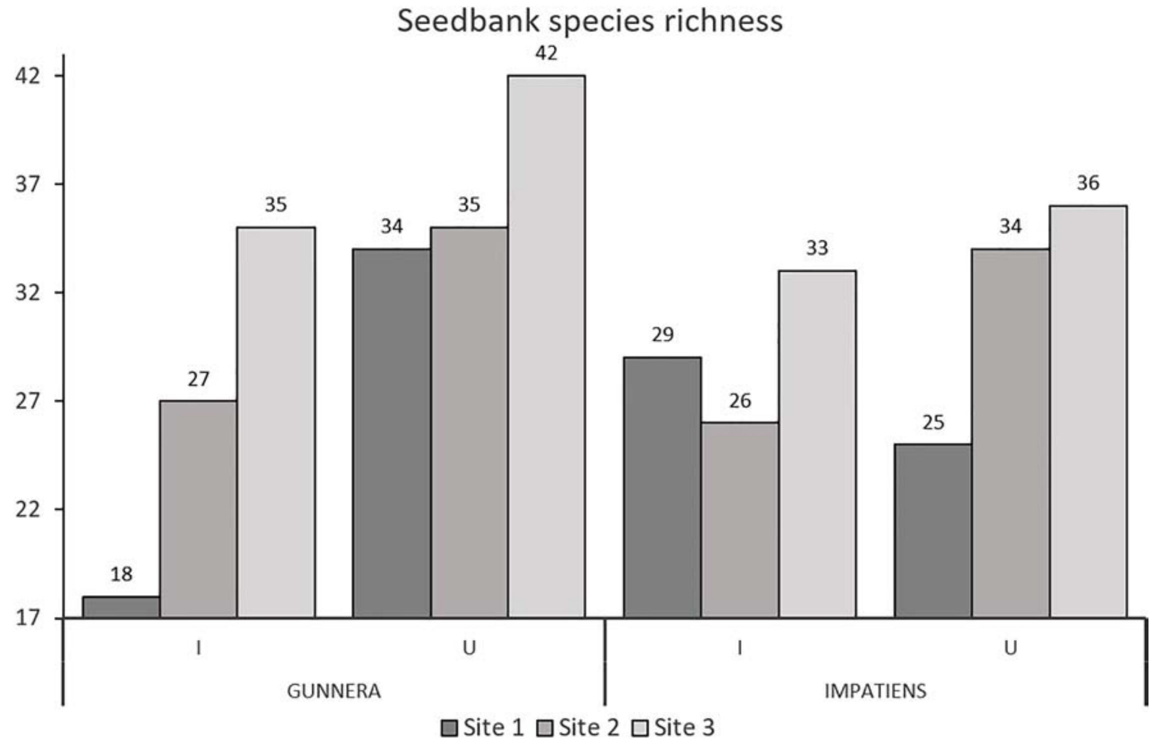

**Figure 3.4. Total number of species found in the soil seed bank of plots invaded by *G. tinctoria* and *I. glandulifera* and comparable uninvaded plots at three sites (S1–S3). I, invaded; U, uninvaded.**

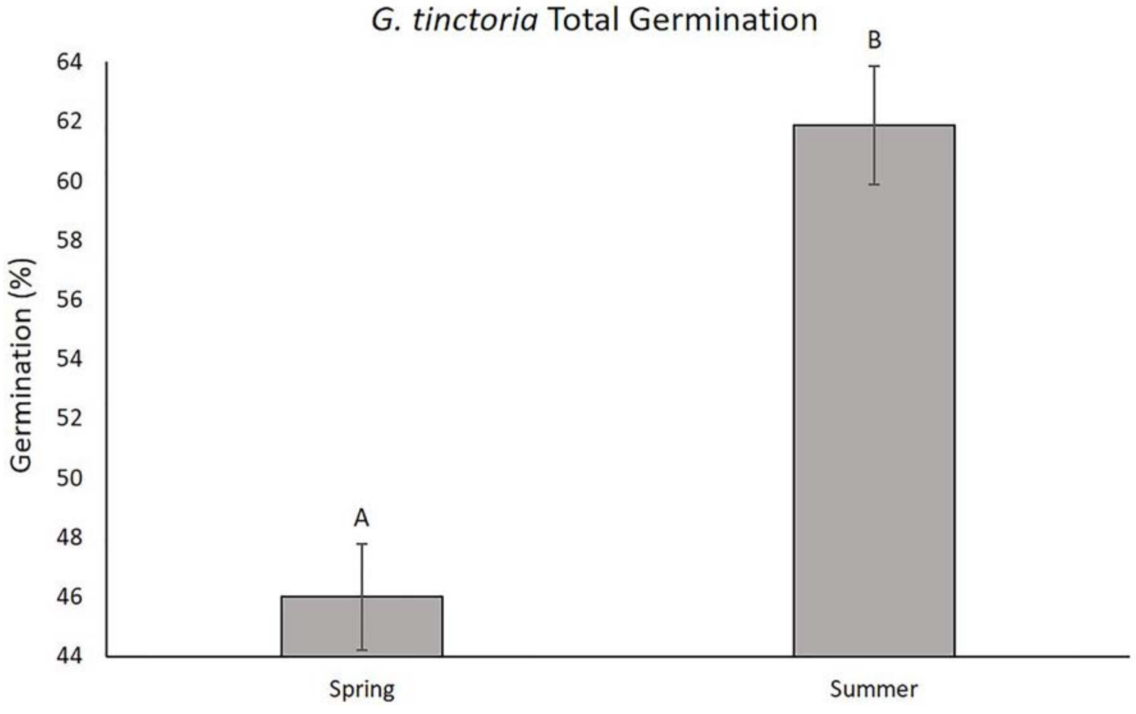

**Figure 3.5. Total germination of *G. tinctoria* seeds exposed to spring (15°C:12°C, day:night) and summer temperatures (20°C:17°C, day:night), averaged across light treatments. Different capital letters indicate significant differences between temperatures ($p$ < 0.05).**

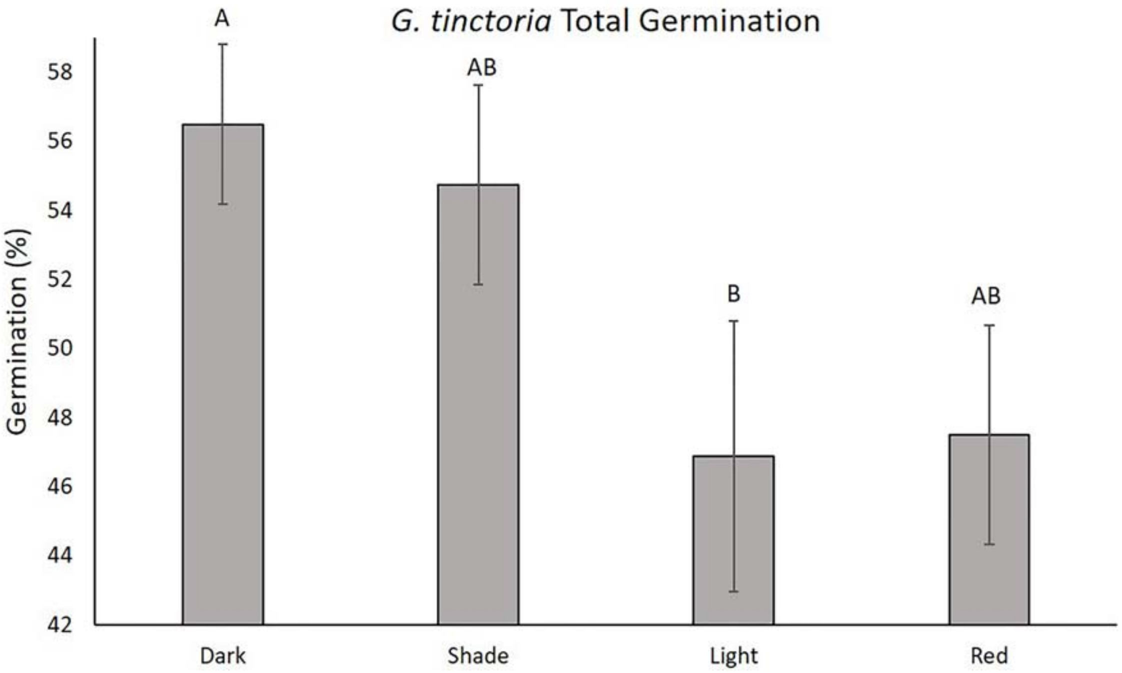

**Figure 3.6. Total germination of *G. tinctoria* seeds in the dark, shade (50% light), full light and red light, averaged across spring and summer temperatures. Different capital letters indicate significant differences between light regimes ($p < 0.05$).**

**Table 3.4. Results of a PERMANOVA testing the effects of temperature (T) and light (I) on total germination and the speed of germination of *G. tinctoria***

| Source of variation | Total germination (*p*) | Germination speed (*p*) |
|---|---|---|
| Temperature (T) | 0.0001 | 0.0001 |
| Light (I) | 0.0405 | 0.0163 |
| T × I | 0.3824 | 0.4352 |

**Values where $p < 0.05$ are significant.**

(20.9 ± 2.67 mg $NO_3^-$ $kg^{-1}$ soil; Figure 3.12a) than in uninvaded areas, while, in contrast, soil $NH_4$ levels were lower in invaded areas (1.78 ± 0.2 mg $NH_4$ $kg^{-1}$ soil) than in uninvaded areas (7.2 ± 0.8 mg $NH_4$ $kg^{-1}$ soil). No significant differences were observed between invaded and uninvaded areas at the River Lagan sites. Soil $NO_3^-$ levels at depths of 0–5 cm were not significantly different between invaded *I. glandulifera* sites (0.7 ± 0.4 mg $NO_3^-$ $kg^{-1}$ soil) and adjacent uninvaded communities (1.8 ± 0.9 mg $NO_3^-$ $kg^{-1}$ soil) ($p = 0.17$; Figure 3.12b), although there are indications that $NO_3^-$ levels were higher in uninvaded communities. Similarly, soil $NH_4$ levels were not significantly different between invaded and uninvaded River Lagan sites (Figure 3.12b, lower panel). Similar results were obtained in January 2018 for soil $NO_3^-$ and $NH_4$ concentrations measured across the two study sites (Figure 3.13a and b). These results suggest that invasive species can have very different effects on soil nitrogen availability and that the direction and magnitude of these effects depend on the “identity” of the invader. For example, high $NO_3^-$ levels in areas invaded by *G. tinctoria* could be related to the fact that this species has a symbiotic relationship with a nitrogen-fixing cyanobacterium or could be due to higher litter production, which may contribute to a higher nitrogen input to the soil, and/or to a higher soil pH. However, there was no evidence that invasions increased total soil nitrogen concentrations. The opposite trend was shown for soil $NH_4$ levels, and this could be attributable to differences in nitrification processes compared with native communities, something that requires further investigation.

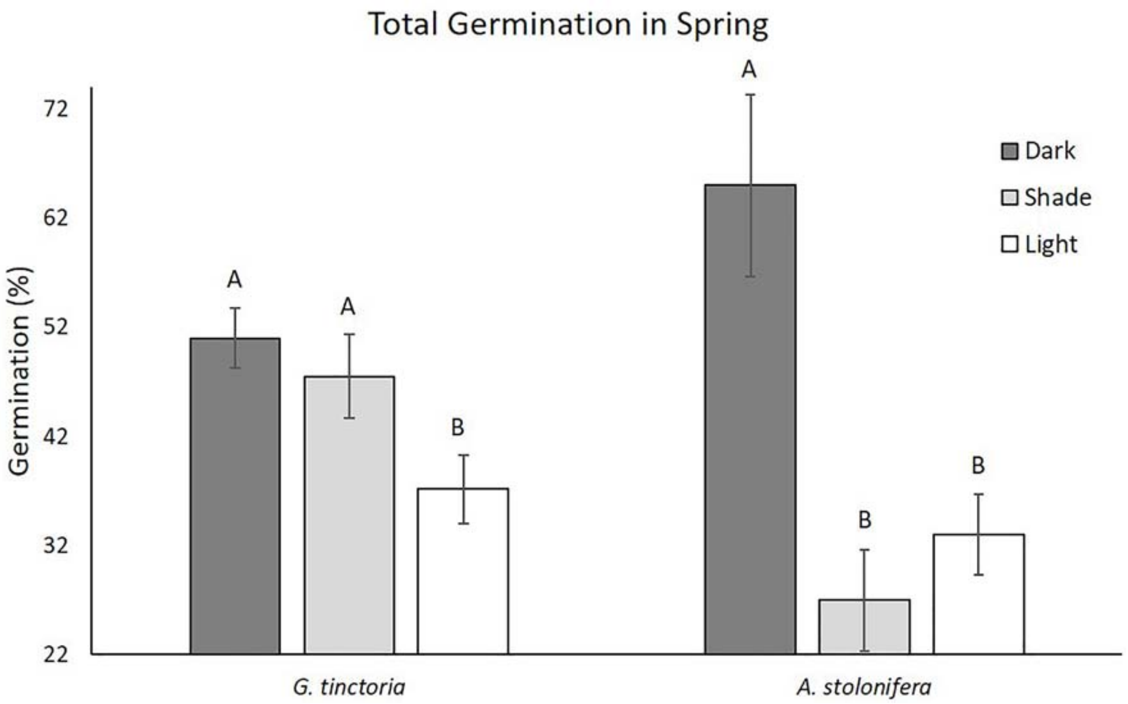

**Figure 3.7. Germination of *G. tinctoria* and *A. stolonifera* exposed to spring temperatures (15°C:12°C, day:night), in the dark, shade (50% light) and full light. Different capital letters indicate significant differences between light regimes ($p < 0.05$).**

We found that the soil carbon concentration was significantly higher in the native community (17.8% ± 0.9%) than in areas invaded by *G. tinctoria* (13.3% ± 0.7%; $p < 0.0001$; Figure 3.14a). Soil carbon did not significantly change with soil depth (0–5 vs 5–20 cm depth; Figure 3.14a). Soil carbon stocks (0–20 cm depth) were also significantly higher in the native plant community (121 ± 5.9 Mg C ha$^{-1}$, $p < 0.0001$)

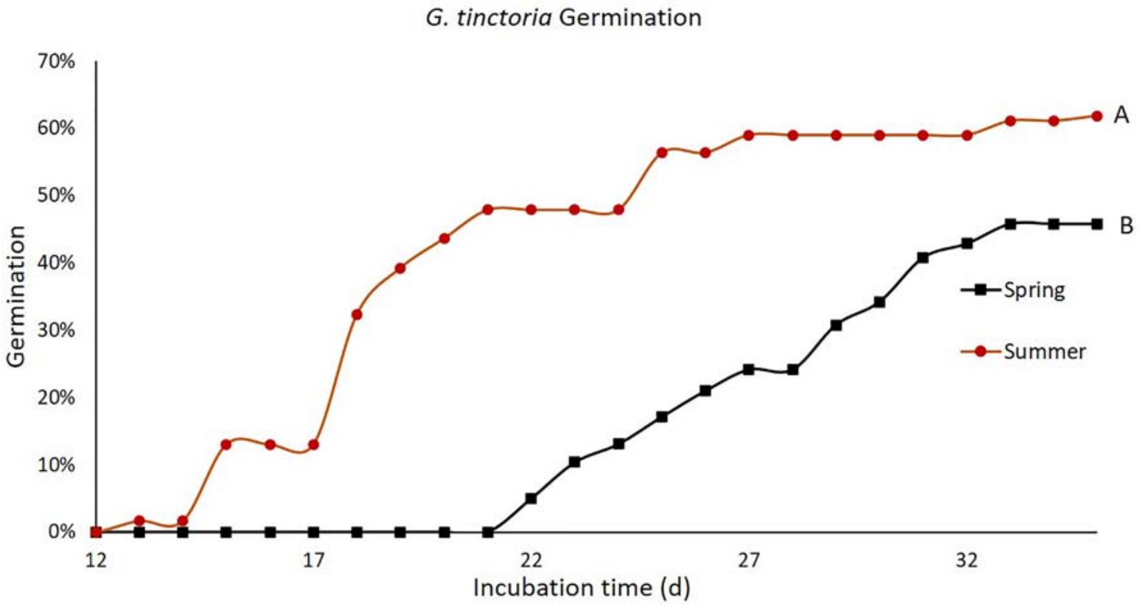

**Figure 3.8. Total germination of *G. tinctoria* during 35 days of exposure to spring (15°C:12°C, day:night) and summer temperatures (20°C:17°C, day:night), averaged across light treatments. Different capital letters indicate significant differences in the speed of germination between temperatures ($p < 0.05$).**

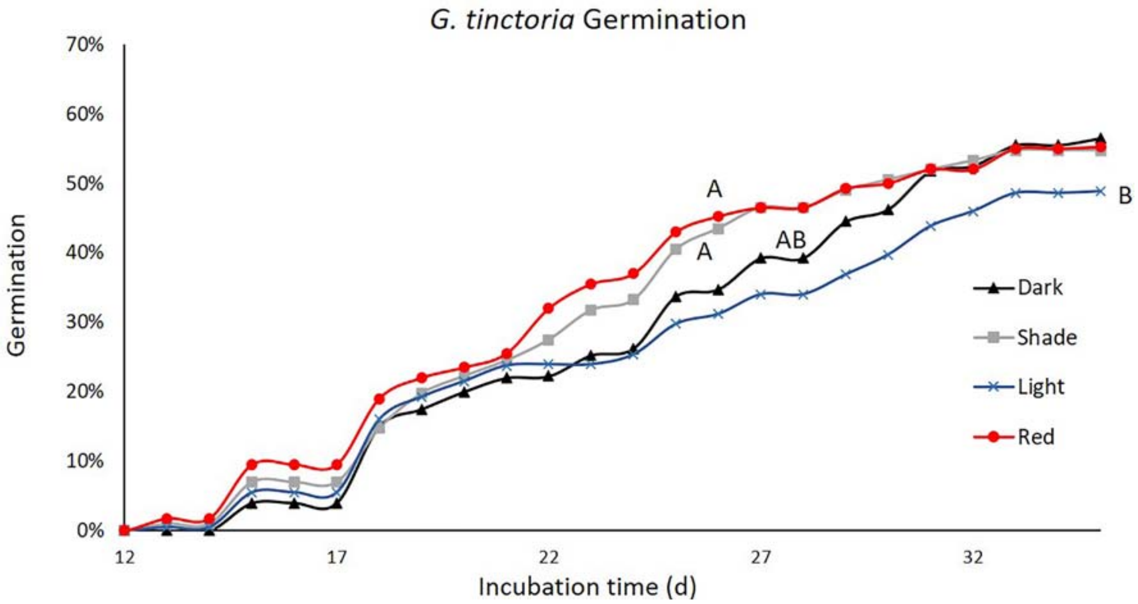

**Figure 3.9. Total germination of *G. tinctoria* seeds during 35 days of exposure to darkness, shade (50% light), full light and red light, averaged across spring and summer temperatures. Different capital letters indicate significant differences in germination speed between light regimes ($p < 0.05$).**

**Table 3.5. Results of a PERMANOVA testing the effects of species (Sp) and light (l) on total germination and germination speed of *G. tinctoria* and *A. stolonifera* exposed to spring temperatures**

| Source of variation | Total germination (*p*) | Germination speed (*p*) |
|---|---|---|
| Species (Sp) | 0.5781 | 0.0375 |
| Light (l) | 0.0006 | 0.0003 |
| Sp × l | 0.0016 | 0.0001 |
| l (*G. tinctoria*) | 0.017 | 0.023 |
| l (*A. stolonifera*) | 0.0032 | 0.0043 |

than in areas invaded by *G. tinctoria* (84 ± 3.7 Mg C ha$^{-1}$), mainly because carbon stocks are higher at depths between 5 and 20 cm (Figure 3.14b). Soil nitrogen was also significantly higher in the native community (1.12% ± 0.05%) than in areas invaded by *G. tinctoria* (0.89% ± 0.04%; $p = 0.0003$). Similar findings were noted for the soil nitrogen stocks, which were significantly higher in the native plant community (9.96 ± 0.89 Mg N ha$^{-1}$) than in areas invaded by *G. tinctoria* (6.44 ± 0.8 Mg N ha$^{-1}$).

No differences in soil carbon concentration (averaged over depths of 0–20 cm) were observed between plant communities invaded by *I. glandulifera* (5.2% ± 0.3%) and adjacent native communities (4.7% ± 0.3%; Figure 3.15a). Nor were carbon stocks significantly different in soil underlying the two types of plant community (Figure 3.15b). However, soil nitrogen concentration was significantly higher under *I. glandulifera* (0.37% ± 0.01%) than under native communities (0.3% ± 0.01%; $p = 0.01$), as was soil nitrogen stock (1.56 ± 0.1 Mg N ha$^{-1}$ compared with 1.29 ± 0.1 Mg N ha$^{-1}$; $p = 0.03$).

To summarise these findings, we found that in Achill Island sites, soil carbon and nitrogen concentrations (%), as well as soil carbon and nitrogen stocks (Mg ha$^{-1}$), were significantly higher under uninvaded communities than under communities invaded by *G. tinctoria*. In the River Lagan sites, soil carbon and nitrogen (%) concentrations, as well as soil carbon stocks (Mg ha$^{-1}$), were not significantly affected by *I. glandulifera* invasion, but soil nitrogen stocks were significantly higher in invaded communities than in uninvaded ones. Changes in soil carbon and nitrogen stocks across invaded and native communities may reflect the legacy of longer-term changes in land use and/or the frequency of disturbance events and may

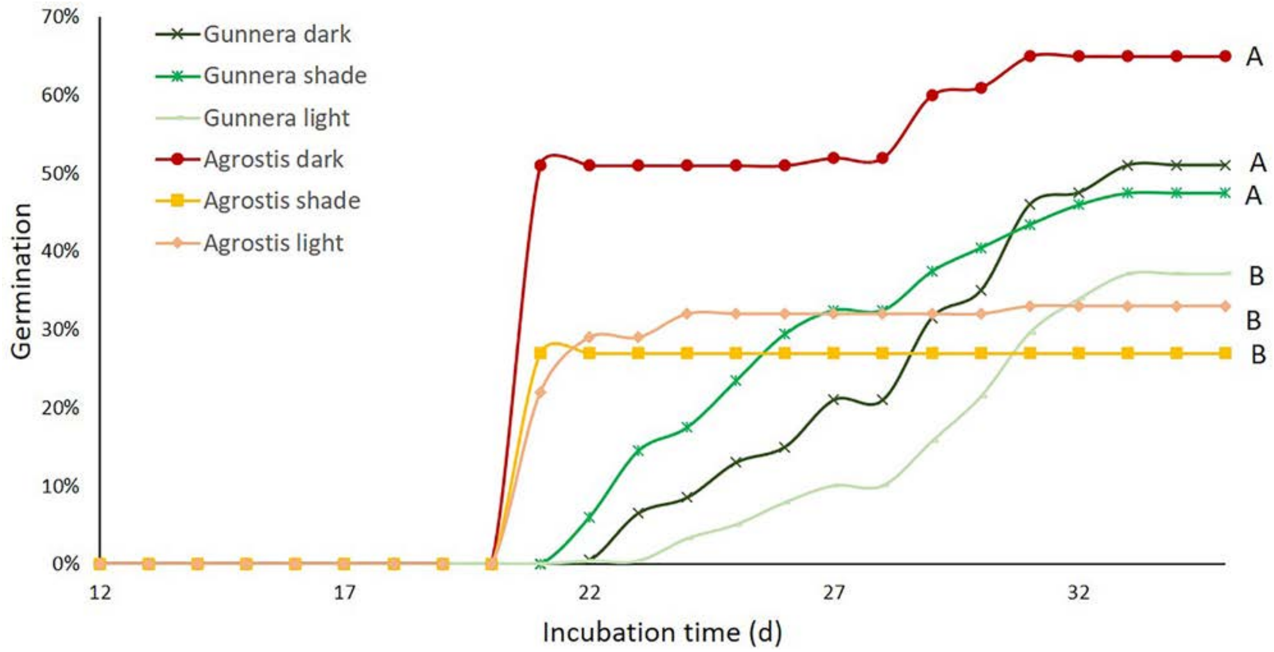

**Figure 3.10. Germination of *G. tinctoria* and *A. stolonifera* during 35 days of exposure to spring temperatures (15°C:12°C, day:night) and two light levels and darkness. Different capital letters indicate significant differences in the speed of germination under different light treatments, for each species separately ($p$<0.05).**

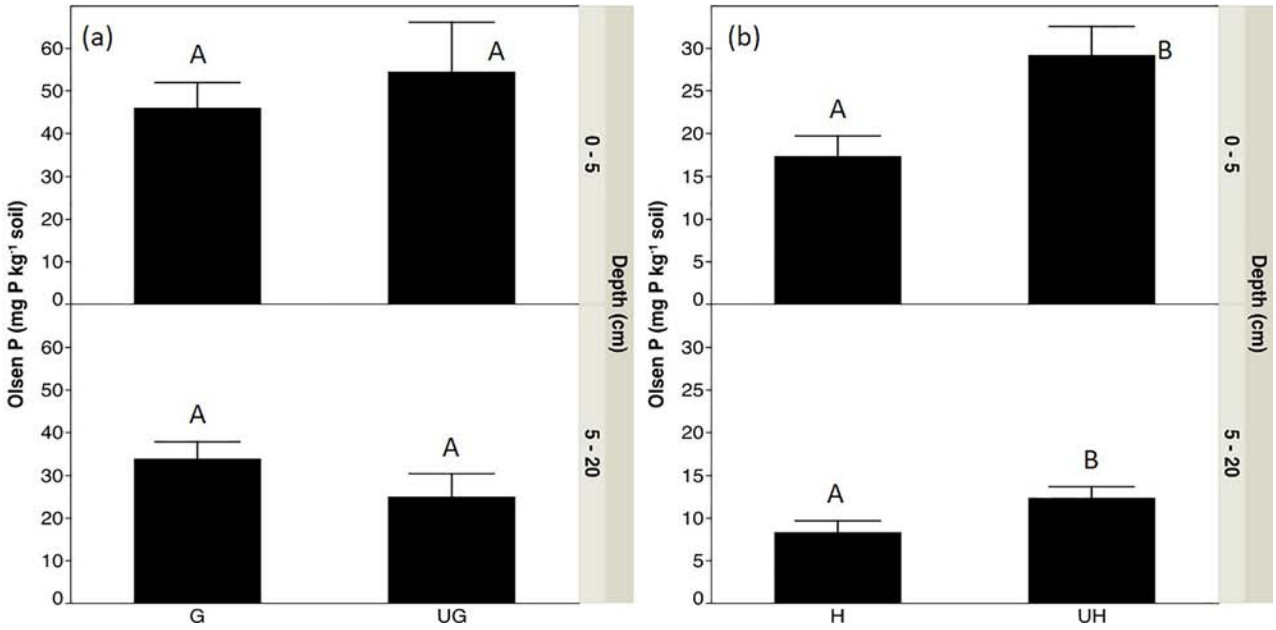

**Figure 3.11. Soil phosphorus measured in January 2018 (a) on Achill Island and (b) at the River Lagan sites at two soil depths. G, community invaded by *G. tinctoria*; H, community invaded by *I. glandulifera*; UG, comparable uninvaded area (not invaded by *G. tinctoria)*; UH, comparable uninvaded area (not invaded by *I. glandulifera)*. Different capital letters indicate significant differences between invaded and uninvaded communities ($p$<0.05).**

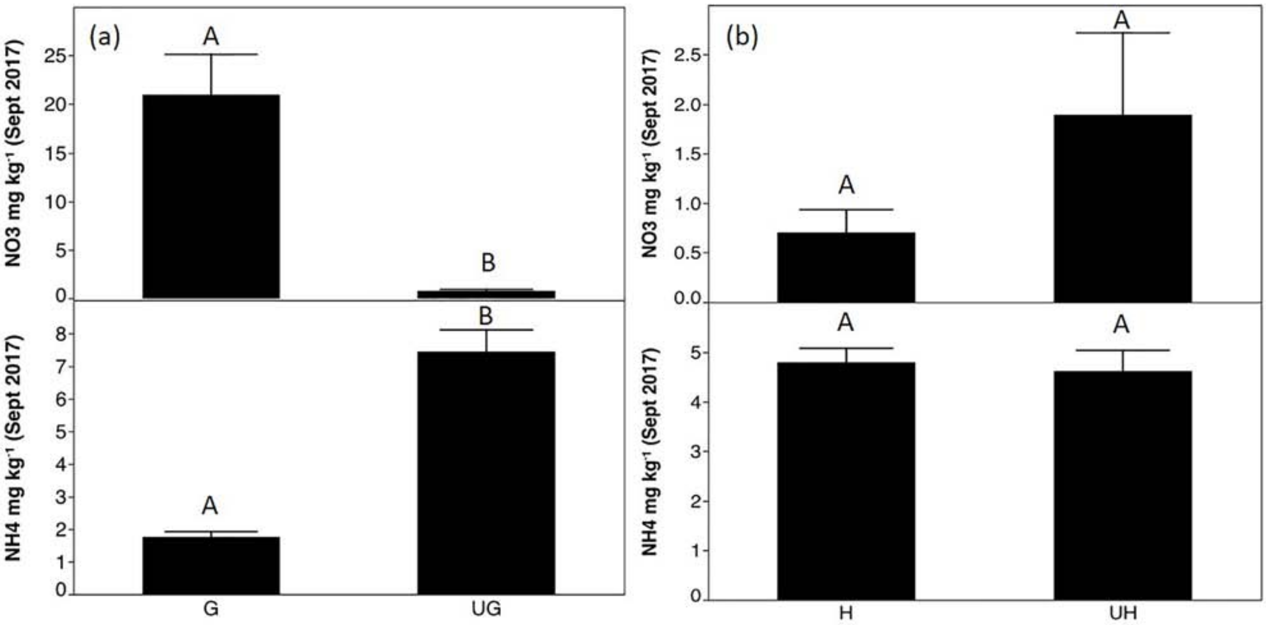

**Figure 3.12. Soil $NO_3^-$ and $NH_4$ (0–5 cm depth) measured in September 2017 (a) on Achill Island and (b) at River Lagan sites. G, community invaded by *G. tinctoria*; H, community invaded by *I. glandulifera*; UG, comparable uninvaded area (not invaded by *G. tinctoria*); UH, comparable uninvaded area (not invaded by *I. glandulifera*). Different capital letters indicate significant differences between invaded and uninvaded communities ($p < 0.05$).**

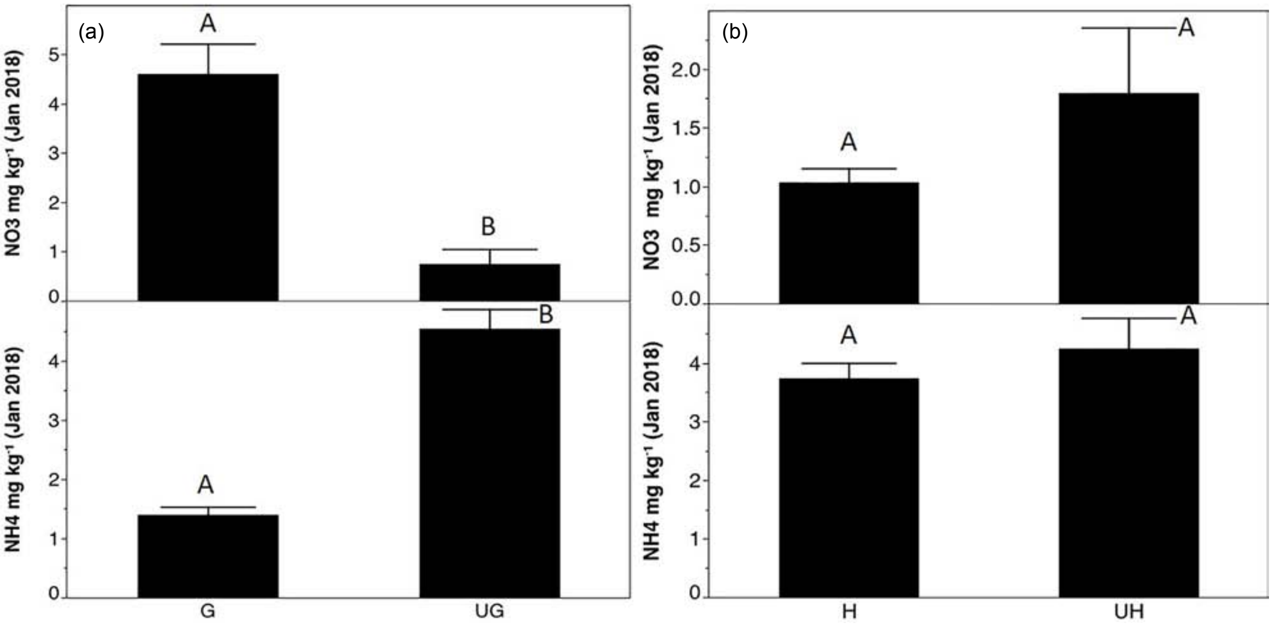

**Figure 3.13. Soil $NO_3^-$ and $NH_4$ (0–5 cm depth) measured in January 2018 (a) on Achill Island and (b) at River Lagan sites. G, community invaded by *G. tinctoria*; H, community invaded by *I. glandulifera*; UG, comparable uninvaded area (not invaded by *G. tinctoria*); UH, comparable uninvaded area (not invaded by *I. glandulifera*). Different capital letters indicate significant differences between invaded and uninvaded communities ($p < 0.05$).**

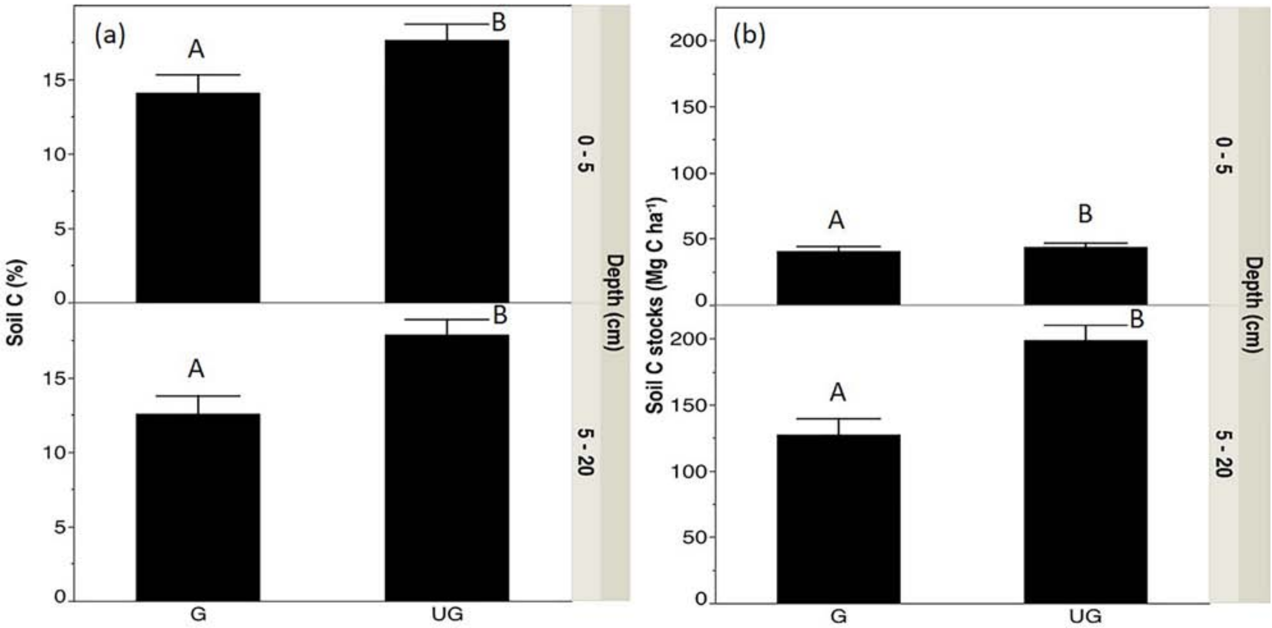

**Figure 3.14. (a) Soil carbon concentration and (b) soil carbon stocks, at different soil depths at Achill Island sites. G, community invaded by *G. tinctoria*; UG, comparable uninvaded area (not invaded by *G. tinctoria*). Different capital letters indicate significant differences between invaded and uninvaded communities ($p < 0.05$).**

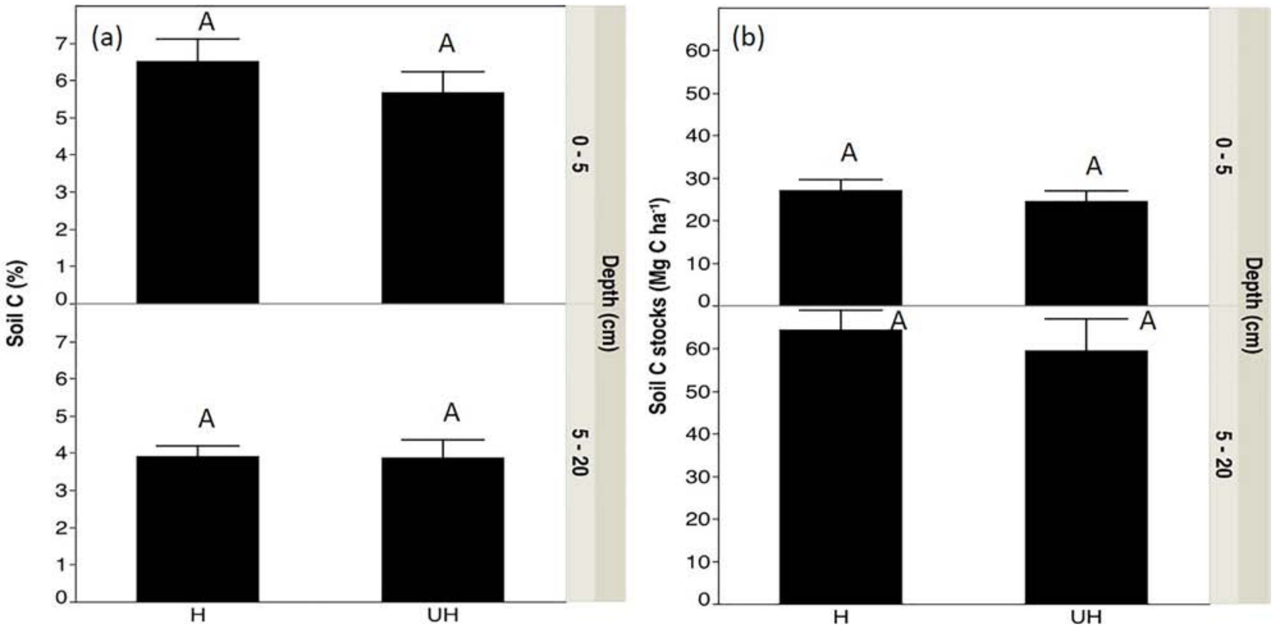

**Figure 3.15. (a) Soil carbon concentration and (b) soil carbon stocks, at different soil depths at the River Lagan sites. H, community invaded by *I. glandulifera*; UH, comparable uninvaded area (not invaded by *I. glandulifera*). Different capital letters indicate significant differences between invaded and uninvaded communities ($p < 0.05$).**

not necessarily be related to the impact of the invasive species. The evidence suggests that *G. tinctoria* may have been present at this location on Achill Island for over 50 years (Fennell *et al*., 2010; Mantoani *et al*., 2020), suggesting that any changes are largely due to the presence of the invader. The date of *I. glandulifera* invasion of the River Lagan sites is unknown. Another important factor to consider is that at both study locations, invaded and uninvaded, the sites have been subjected to human activities/disturbance events for thousands of years and have possibly been affected by agricultural practices (Achill sites) and river management and flooding (Lagan site), resulting in nutrient enrichment, which can affect soil organic matter formation and soil carbon content and stocks.

Soil pH was significantly higher in sites invaded by *G. tinctoria* (pH 6.73) than in uninvaded areas (pH 6.15; $p < 0.0001$). However, soil pH was not significantly different between areas invaded by *I. glandulifera* (pH 5.63) and uninvaded areas (pH 5.6). The statistical analyses suggest that differences in soil $NO_3^-$ and $NH_4$ between the community invaded by *G. tinctoria* and the native community on Achill Island could be partly due to changes in soil pH. For example, soil $NH_4$ levels are negatively related to soil pH, and this might explain why the soil $NH_4$ concentrations were higher in the more acidic (lower pH) soil associated with the native community than in the soil below the invaded community. Differences in soil pH could also explain in part why the native community is associated with higher soil carbon and nitrogen stocks than the community invaded by *G. tinctoria* on Achill Island. The more acidic soil conditions, i.e. lower pH, under the native community could be responsible for reducing the decomposition of leaf and root organic debris and thus increase the accumulation of soil organic matter. Examination of the soil at the River Lagan sites associated with *I. glandulifera* showed that there was a significant relationship between soil pH and soil carbon, and in the opposite direction to that found on Achill Island. For example, soil carbon stocks were higher at high soil pH than at low pH. However, there was no relationship between soil pH and soil $NH_4$ levels. There were no significant effects of invasion on soil microbial biomass on Achill Island (Figure 3.16a; $p = 0.89$) or at the River Lagan sites (Figure 3.16b; $p = 0.89$), suggesting that any modifications are related to differences in the soil microbial communities, not their abundance. However, soil microbial biomass was greater on Achill Island than at the River Lagan sites, probably reflecting the differences in soil organic

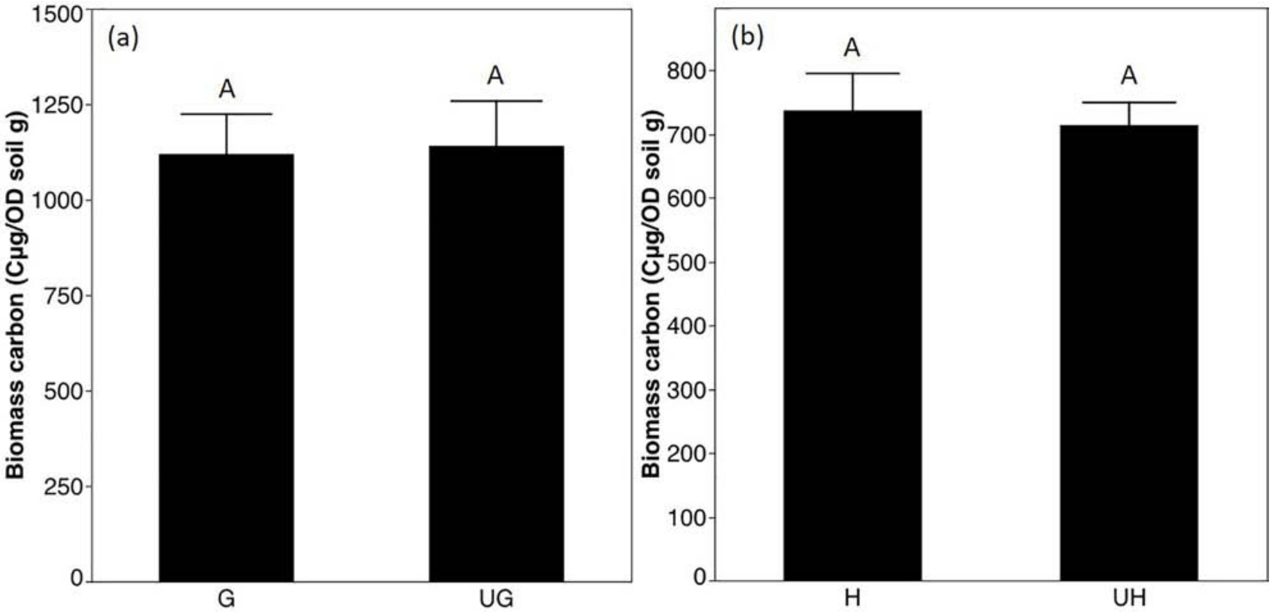

**Figure 3.16. Soil microbial biomass (biomass carbon) (a) on Achill Island and (b) at the River Lagan sites. G, community invaded by *G. tinctoria*; H, community invaded by *I. glandulifera*; UG, comparable uninvaded area (not invaded by *G. tinctoria*); UH, comparable uninvaded area (not invaded by *I. glandulifera*). Different capital letters indicate significant differences between invaded and uninvaded communities ($p < 0.05$).**

carbon concentration and stocks between the two sites.

We found no significant effects of invasion (Figure 3.17a; $p = 0.96$) on the activity of the extracellular enzyme β-1,4-glucosidase (BG) on Achill Island. However, there were positive effects of *I. glandulifera* on BG activity at the River Lagan site (Figure 3.17b; $p = 0.02$) when compared with the native community. This might be because in sites invaded by *I. glandulifera* carbon was relatively less available than nitrogen for microbial metabolism, and thus microbes produce more BG to acquire carbon from soils to sustain their metabolism. In general, BG activity was lowest in the non-invaded River Lagan site, followed by invaded and non-invaded Achill Island sites, with invaded River Lagan sites having the highest BG activity. Overall, however, microbial carbon biomass in soil was higher in the Achill Island sites than in the River Lagan sites.

Overall, the results suggest that the presence of invasive species could have a significant impact on soil ecosystem functioning. For example, the study found a strong positive effect of *Gunnera tinctoria* on soil $NO_3^-$ levels. However, changes in other soil parameters and properties could not be explained by the presence of the invasive species. Furthermore, the study findings indicated that the two invaders differ substantially in their impact at the ecosystem level, suggesting that ecosystem-level impacts will ultimately depend on the identity of the invasive plant species and its interaction with multiple environmental factors and with the previous management history of the site that has been invaded.

## 3.4 Work Package 4

### 3.4.1 Competition experiment

The probability of survival (hereafter survival) decreased rapidly with increasing competition in the fertilised treatments while only the highest competition treatment resulted in a significant reduction in seedling survival in the unfertilised treatments (Figure 3.18). Consequently, there was a significant competition × fertilisation interaction ($p = 0.03$) effect on survival. Remarkably, all the *G. tinctoria* seedlings growing without any competition survived. These results demonstrate the strong negative effect of competition on seedling survival, which becomes even more pronounced under high-fertility conditions. This negative effect, in addition to the very low survival

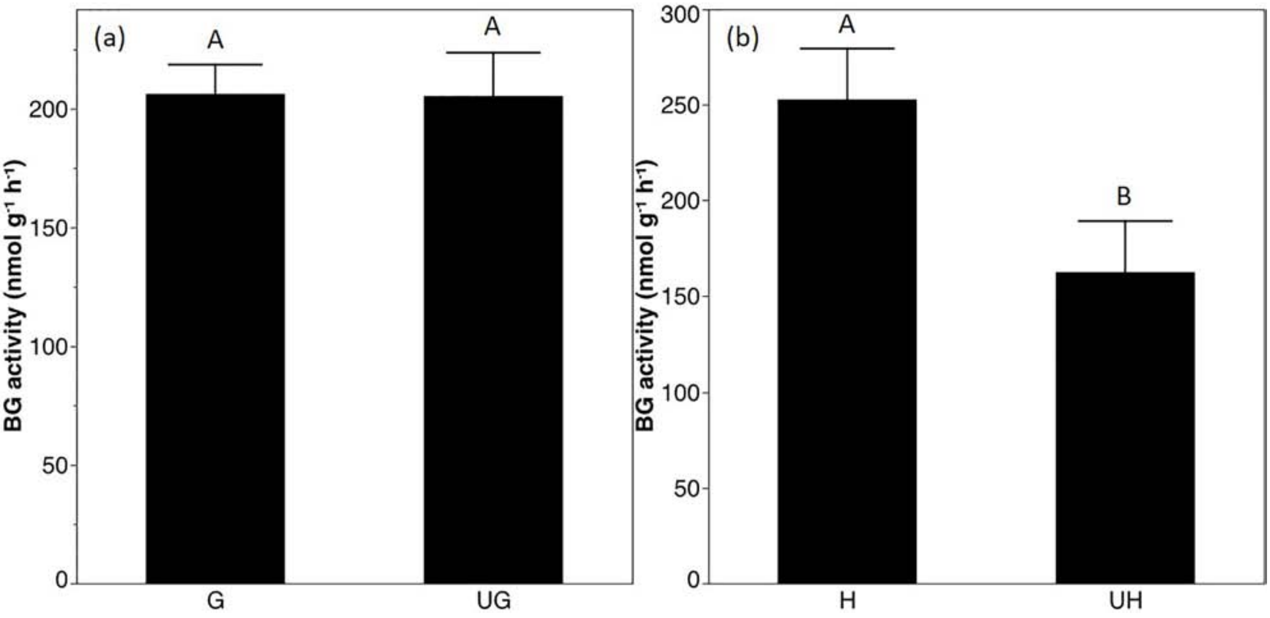

**Figure 3.17. Activity of the extracellular enzyme β-1,4-glucosidase (BG) in soils collected from (a) Achill Island and (b) River Lagan sites. G, community invaded by *G. tinctoria*; H, community invaded by *I. glandulifera*; UG, comparable uninvaded area (not invaded by *G. tinctoria*); UH, comparable uninvaded area (not invaded by *I. glandulifera*). Different capital letters indicate significant differences between invaded and uninvaded communities ($p < 0.05$).**

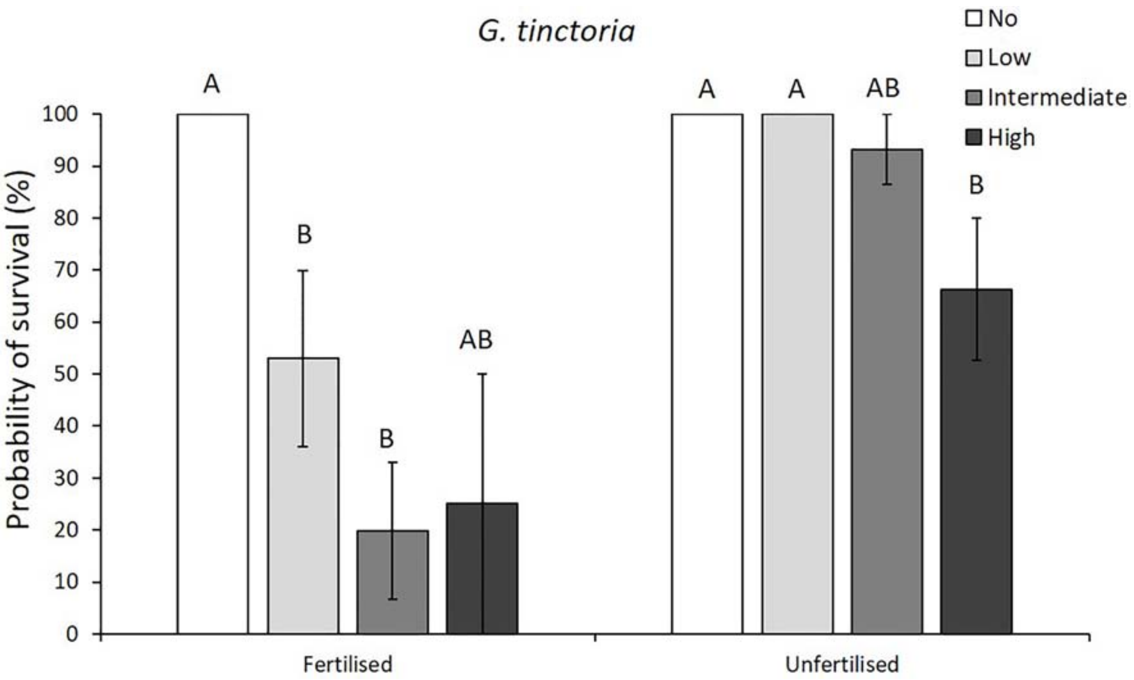

**Figure 3.18. Probability of seedling survival (%) during the last 2 months of the experiment at four competition levels, either fertilised or unfertilised treatments. Different capital letters indicate significant differences between competition levels in fertilised and unfertilised treatments ($p < 0.05$).**

of *G. tinctoria* seedlings observed during the early stages of the experiment, challenges the notion that all invaders are good competitors.

Total biomass of *G. tinctoria* seedlings was greatly reduced by competition, in both fertilised and unfertilised conditions (Figure 3.19a). The negative effect of competition on the biomass of *G. tinctoria* was more pronounced in the fertilised treatments, mainly because the biomass of the fertilised seedlings growing without any competition was sixfold higher than that of the comparable unfertilised treatment. In contrast to this significant increase ($p = 0.03$), fertilisation had no effect on the biomass of seedlings growing in low- and high-competition treatments, while it significantly decreased the biomass of seedlings growing in the intermediate treatment ($p = 0.04$), and thus a significant competition × fertilisation interaction was observed. These results demonstrate that even the lowest level of competition can significantly reduce *G. tinctoria* standing biomass, especially when resources are widely available. Hence, *G. tinctoria* might not be a good competitor, particularly under fertile conditions, as its performance is greatly reduced even under a low competitive pressure. The large difference in biomass between fertilised and unfertilised treatments when *G. tinctoria* was growing alone might not be directly related to the availability of soil nitrogen, as *G. tinctoria* is thought to rely almost solely on fixed nitrogen (Osborne *et al*., 1992). Either this species can utilise some soil nitrogen, particularly if resources are high as indicated earlier, or it is an effect related to the increased availability of other nutrients, such as phosphorus and potassium, as a consequence of fertilisation.

Leaf, root and rhizome biomass of *G. tinctoria* followed the same pattern as the total biomass, with only one exception, in the unfertilised treatments. There, seedlings grown without competition did not have significantly more biomass than seedlings grown in the low-competition treatment. Petiole biomass also followed the same pattern as the other plant parts, with the single exception of the unfertilised treatments. In these treatments, seedlings growing at the intermediate competition level did not have a significantly higher petiole biomass than seedlings grown at the high-competition level.

Both leaf number and leaf area were significantly reduced by competition in both fertilised and unfertilised treatments (Figure 3.19b and c) although the average decreases were much greater under fertilised conditions (82% decrease in leaf number and 97% decrease in leaf area) than under unfertilised

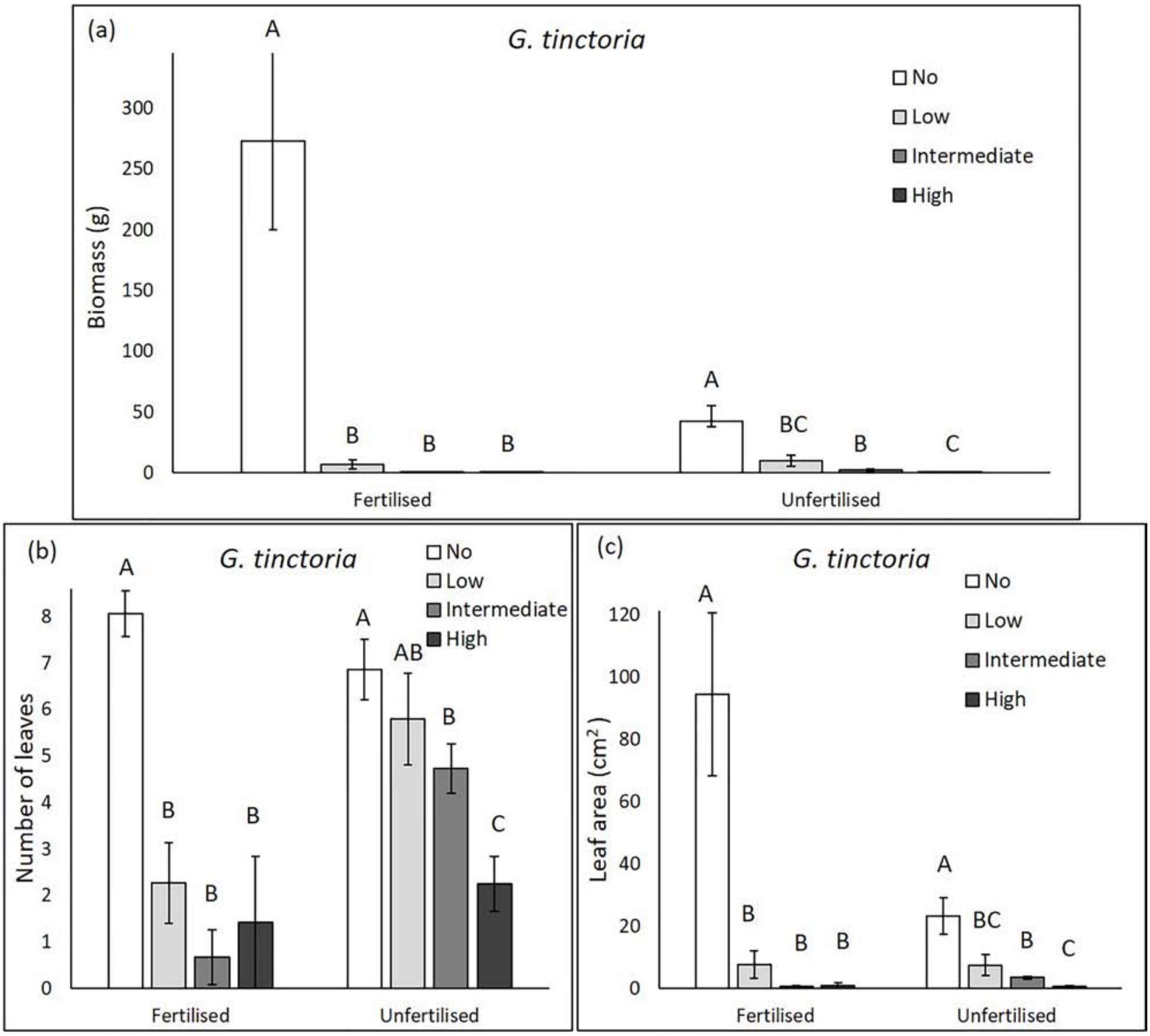

**Figure 3.19. (a) Total seedling biomass (g), (b) average number of leaves per seedling and (c) average leaf area (cm$^2$) for four competition levels, in fertilised and unfertilised treatments. Different capital letters indicate significant differences between competition levels in fertilised and unfertilised treatments ($p < 0.05$).**

conditions (corresponding values of 38% and 84%, respectively). However, in fertilised treatments, both leaf number and leaf area were significantly reduced at all competition levels, while in the unfertilised treatments, although leaf area was significantly reduced by all competition treatments, leaf number was reduced only in the intermediate- and high-competition treatments. Consequently, competition reduced *G. tinctoria* leaf biomass by reducing both leaf number and leaf area in fertilised and unfertilised treatments. Fertilisation caused a significant decrease in the number of leaves and leaf area, but only in the case of the intermediate-competition treatment. It had no significant effect in the other competition treatments; hence, a significant fertilisation × competition interaction occurred. In contrast, fertilised seedlings grown without competition had a much larger leaf area than unfertilised seedlings, but this difference was not significant, perhaps because of the high variability (Figure 3.19c). These results further demonstrate the poor performance of *G. tinctoria* even under low competitive pressure, in agreement with seedling survival and the total biomass results, and they show that fertilisation can have a positive or negative effect on invasion, depending on the competitive pressure applied on the invader by the native community. All the surviving young plants were infected by *N. punctiforme* (Figure 3.20) and the ratio of infected area to total rhizome area was not significantly affected by competition, fertilisation or their interaction.

The biomass of *A. stolonifera* increased significantly under both fertilised and unfertilised conditions, as expected, as the cutting height increased (Figure 3.21), reflecting differences in the amount of biomass removed as a result of the cutting treatments. In contrast to *G. tinctoria*, the biomass of *A. stolonifera* was significantly increased by fertilisation, indicating that *A. stolonifera* was able to take up the additional resources (Figure 3.21).

Nitrogen concentrations in leaves, petioles and rhizome of *G. tinctoria* seedlings was not significantly affected by competition, fertilisation or their interaction, demonstrating that competition did not affect the ability

**Figure 3.20. Rhizomes of young plants of *G. tinctoria* infected by *N. punctiforme*.**

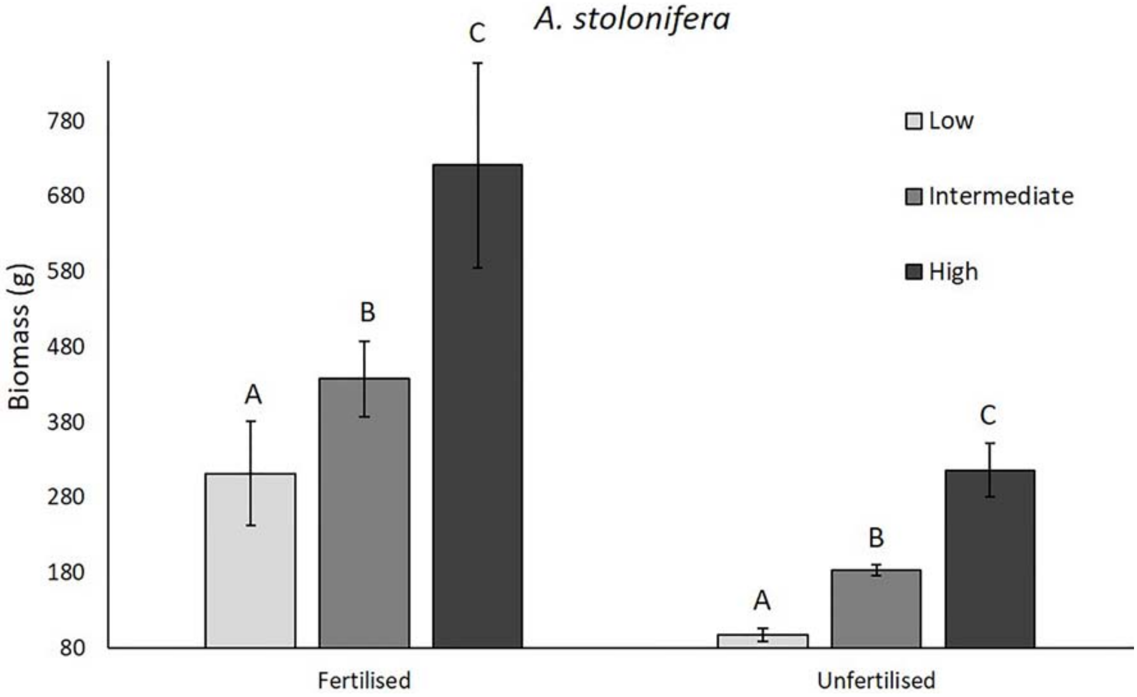

**Figure 3.21. Total biomass of *A. stolonifera* (g) per half-tray at three clipping heights in the fertilised and unfertilised treatments. Different capital letters indicate significant differences between clipping heights in fertilised and unfertilised treatments ($p < 0.05$).**

of G. *tinctoria* to utilise nitrogen (Figure 3.22a and b). However, competition significantly decreased the nitrogen concentration in roots (Figure 3.22d).

Increased competition significantly increased the above-ground tissue nitrogen concentration in *A. stolonifera*, but only under fertilised conditions, and thus a significant competition × fertilisation interaction was observed (Figure 3.23). This difference is unlikely to be caused by the competitive pressure the three *G. tinctoria* seedlings applied to *A. stolonifera*, rather they are related to differences in grass height through the effects of clipping.

Competition significantly decreased the values of the nitrogen isotopic ratio ($\delta^{15}$N) in *G. tinctoria* leaves, but only in fertilised treatments (significant competition × fertilisation interaction) (Figure 3.24). In the fertilised treatments, where *G. tinctoria* was growing without competition, values of $\delta^{15}$N were around +5, while in all other treatment combinations, $\delta^{15}$N was closer to 0, as might be expected, as the $\delta^{15}$N values of plants that rely exclusively on nitrogen fixation are usually ≈ 0, or slightly negative (Handley, 2002; Unkovich, 2013), as shown in a recent study on *G. tinctoria* (Mantoani *et al*., 2020) (Figure 3.24).

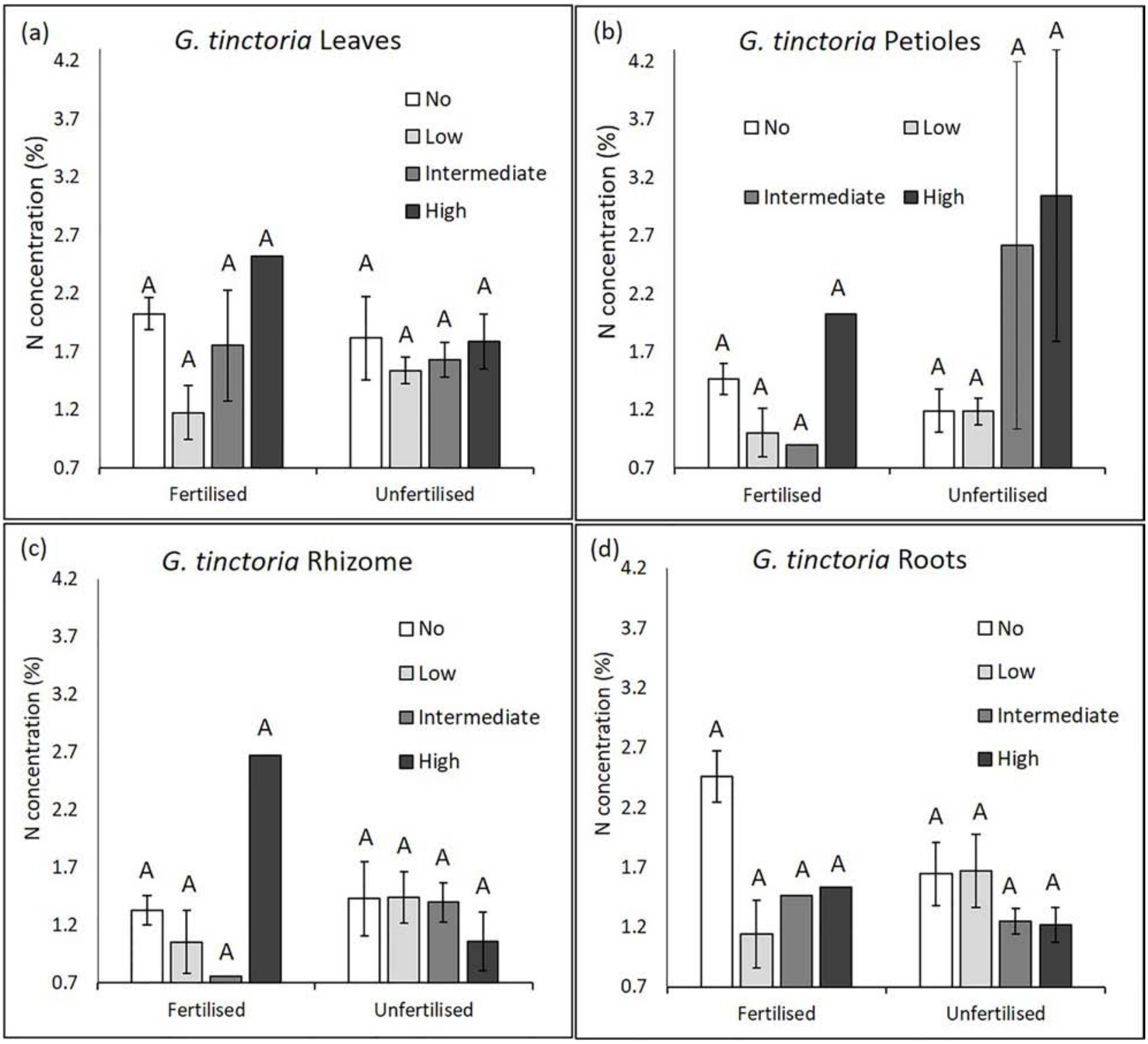

**Figure 3.22. The concentration of nitrogen (%) in (a) leaves, (b) petioles, (c) rhizome and (d) roots of *G. tinctoria* under four competition treatments, in fertilised and unfertilised conditions. Different capital letters indicate significant differences between competition levels in fertilised and unfertilised treatments ($p < 0.05$).**

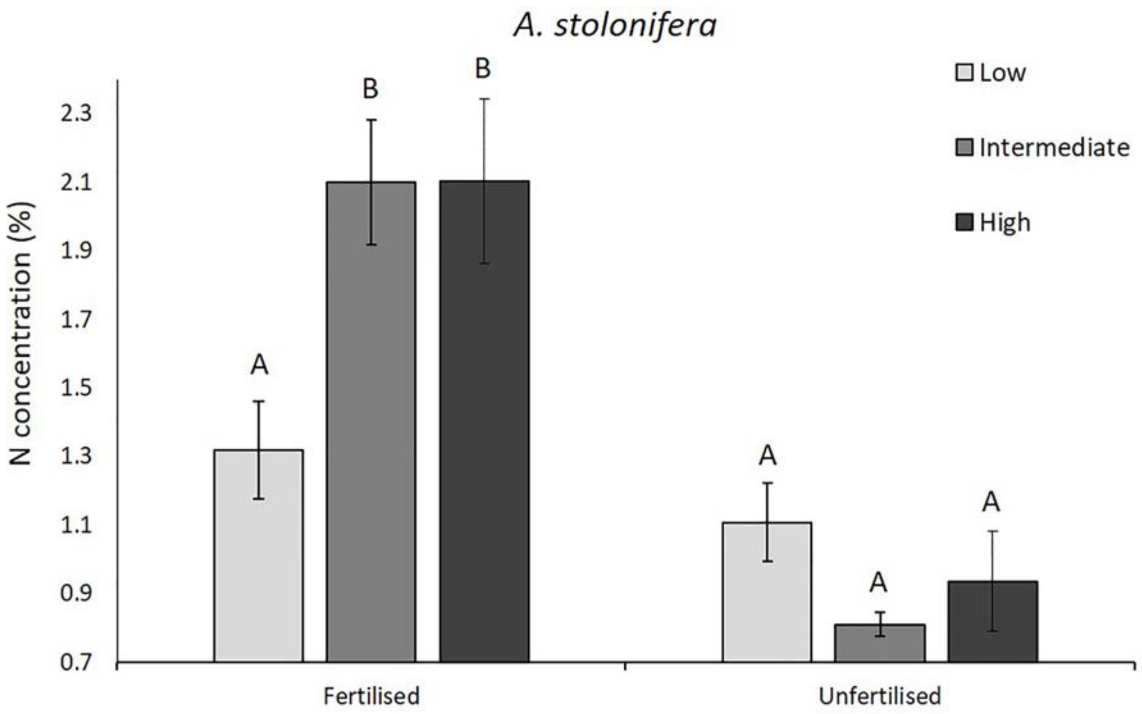

**Figure 3.23. The concentration of nitrogen (%) in leaves of *A. stolonifera* under three clipping levels, in the fertilised and unfertilised treatments. Different capital letters indicate significant differences between clipping heights in fertilised and unfertilised treatments ($p < 0.05$).**

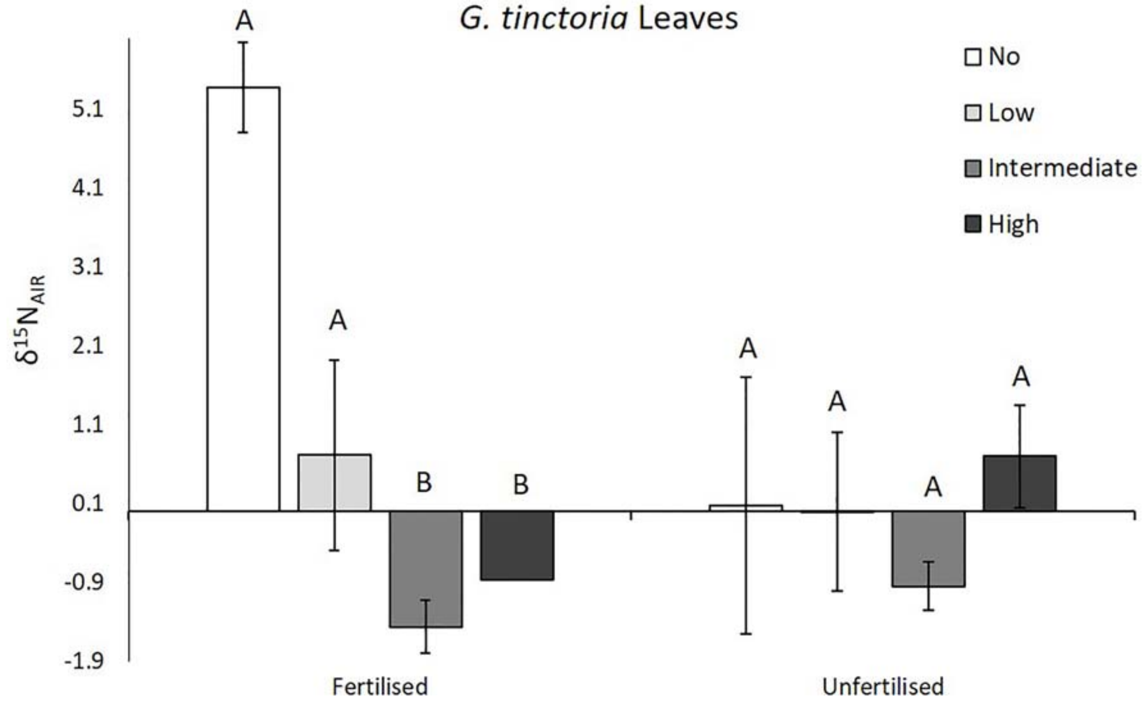

**Figure 3.24. The nitrogen isotopic ratio ($\delta^{15}$N) in leaves of *G. tinctoria* under four competition levels and two fertilisation treatments. Different capital letters indicate significant differences between competition levels in the fertilised and unfertilised treatments ($p < 0.05$).**

Consequently, the result that fertilisation significantly increased the nitrogen isotopic ratio in the treatment without any competition may challenge the notion that *G. tinctoria* always relies on nitrogen fixation. However, positive values for nitrogen isotopes were found in field studies where the soils are nitrogen deficient and where the *G. tinctoria* cyanobionts were still fixing nitrogen, and this was attributed to remobilisation of nitrogen from the rhizome and associated nitrogen transformations, rather than providing evidence that nitrogen was being taken up from the soil (Mantoani *et al*., 2020).

Carbon concentrations in leaves, rhizome and roots were not significantly affected by competition, fertilisation or their interaction. However, competition significantly increased the carbon concentration in the petioles, with *G. tinctoria* seedlings growing under intermediate competition having significantly higher carbon concentration in their petioles than seedlings growing in the low- or no-competition treatments. Carbon concentrations in the leaves of *A. stolonifera* were not affected by competition, fertilisation or their interaction.

In general, values of the stable carbon isotope ratio ($\delta^{13}C$) decreased with the intensity of competition, in agreement with the results of a study that found more negative $\delta^{13}C$ values in the presence of below-ground competition for nitrogen (Williams *et al*., 1991) and perhaps other nutrients that cause variations in photosynthetic capacity, such as phosphorus (Domingues *et al*., 2010; Cernusak *et al*., 2013). This reduction was observed for all plant parts examined (Figure 3.25a–d). The reduction in $\delta^{13}C$ values with competition was much greater in the fertilised treatments (≈5‰) than in the unfertilised ones (≈2‰), indicating a greater competitive effect of fertilised treatments, in agreement with seedling survival and biomass results. Fertilisation significantly increased the values of $\delta^{13}C$ in the below-ground plant parts, independent of the competition treatment (Figure 3.25c and d). Furthermore, fertilisation increased the values of $\delta^{13}C$ in the above-ground plant parts (although not significantly for the leaves) of seedlings grown in the absence of competition (Figure 3.25a and b). This result agrees with previous studies that reported increased $\delta^{13}C$ values after nitrogen addition in some grass species (Tsialtas *et al*., 2005); however, a decrease or no change in $\delta^{13}C$ after nitrogen addition has also been reported in the literature (Handley *et al*., 1994; Shangguan *et al*., 2000; Tsialtas *et al*., 2005). Finally, in *A. stolonifera*, $\delta^{13}C$ was also reduced by an increase in cutting height (Figure 3.26), while it was not affected by fertilisation or competition × fertilisation interactions.

Potassium and phosphorus concentrations in *G. tinctoria* seedlings were both significantly increased by fertilisation and significantly decreased by competition, while the competition × fertilisation interaction was not significant (Figure 3.27a and b). The concentrations of magnesium and sulfur in *G. tinctoria* seedlings were also significantly decreased by fertilisation (Figure 3.28) but were unaffected by competition or by the fertilisation × competition interaction, while the concentration of calcium was not affected by any factor or by any interaction between factors. Micronutrient (sodium, iron, manganese, zinc, cobalt) concentrations in *G. tinctoria* were not affected by competition, fertilisation or their interaction, with the exception of the concentration of sodium, which was significantly increased by competition (Figure 3.29).

A significant competition × fertilisation interaction was observed for all the macronutrient concentrations in *A. stolonifera* tissues, except for phosphorus (Figure 3.30). The phosphorus concentration was increased by fertilisation and decreased by competition, but the fertilisation × competition interaction was not significant. In general, fertilisation resulted in increased potassium concentrations but decreased concentrations of the other three macronutrients. Fertilisation had no significant effect on magnesium or sulfur concentrations under the high-competition conditions. Furthermore, competition had a significant effect on potassium, magnesium and sulfur concentrations only in the unfertilised treatments, causing a significant increase in potassium, but a significant decrease in magnesium and sulfur. Competition increased the calcium concentration under fertilised conditions, but it decreased it under unfertilised conditions. The concentration of iron was significantly decreased by competition, while it was not affected by fertilisation or the interaction between competition and fertilisation. The concentration of manganese was significantly decreased by fertilisation while the concentrations of zinc and cobalt were significantly increased by fertilisation. Manganese, zinc and cobalt were not affected by competition or fertilisation × competition interaction. The concentration of sodium was significantly affected

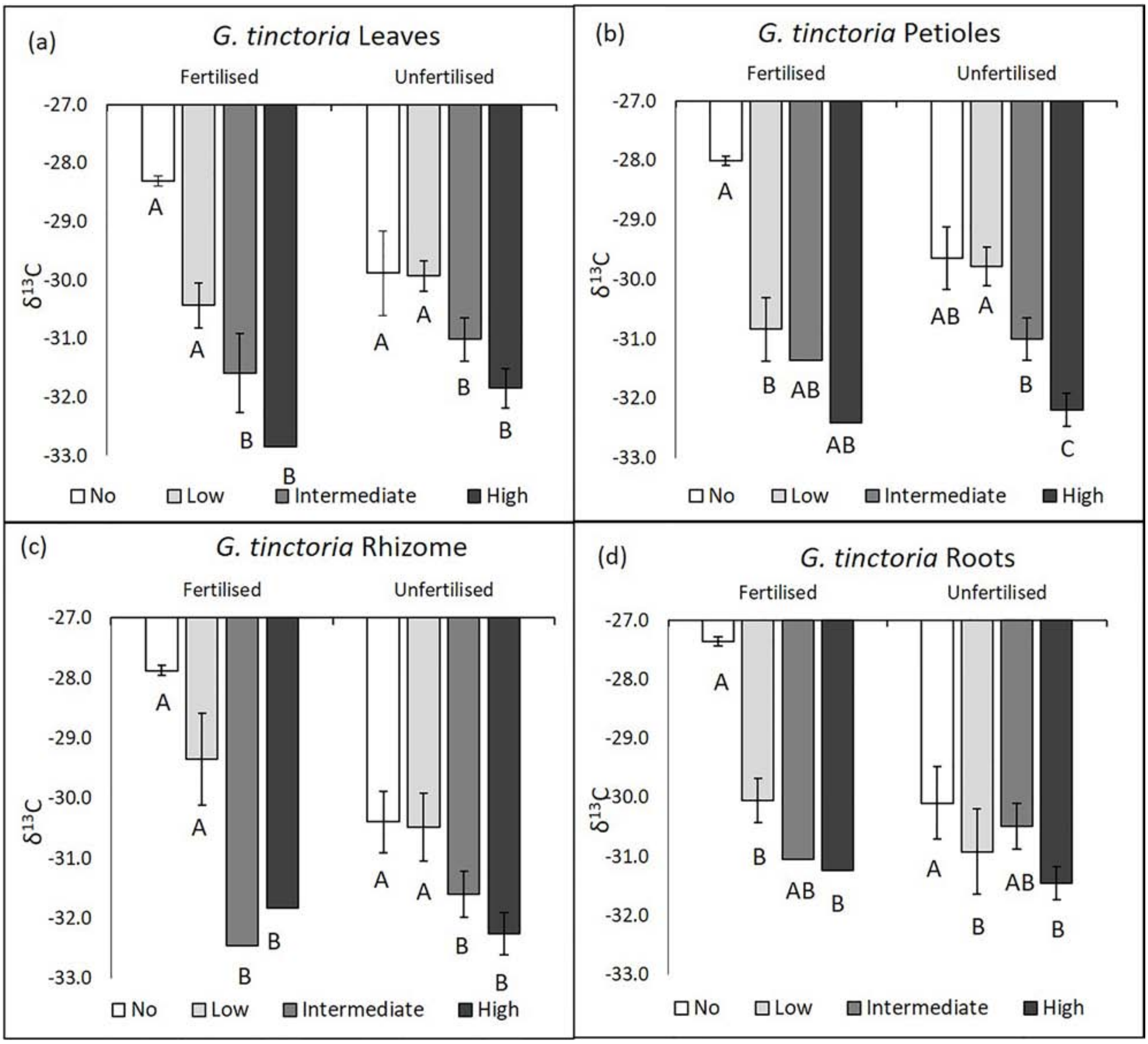

**Figure 3.25. Values for δ$^{13}$C in different plant parts of *G. tinctoria* in the four competition treatments, with and without fertilisation. Different capital letters indicate significant differences between competition levels in the fertilised and unfertilised treatments ($p < 0.05$).**

by fertilisation × competition interaction. Fertilisation decreased sodium concentrations under all treatments, but especially the low-competition treatment. Competition also decreased the sodium concentration, but only in the case of the unfertilised treatments. PERMANOVA *p*-values for biomass, leaf area and number, stable isotope ratios, carbon, macronutrients and micronutrient concentration for *G. tinctoria* and *A. stolonifera* are given in Table 3.6.

### 3.4.2 Phenology experiment

The probability of survival of *G. tinctoria* seedlings was very low, and it was not significantly affected by differences in planting time. By the end of the experiment, only three seedlings had survived, two subjected to the intermediate treatment and one subjected to the late treatment; thus, the biomass or leaf area comparisons are unlikely to be very meaningful. Furthermore, the biomass of *A. stolonifera* was not significantly affected by the sowing time.

The results from this WP show that the performance of *G. tinctoria* was reduced by competition, even under the lowest competitive pressure, based on seedling survival, biomass production, leaf number and leaf area. When *G. tinctoria* was grown under competitive, fertilised conditions, it was able to take

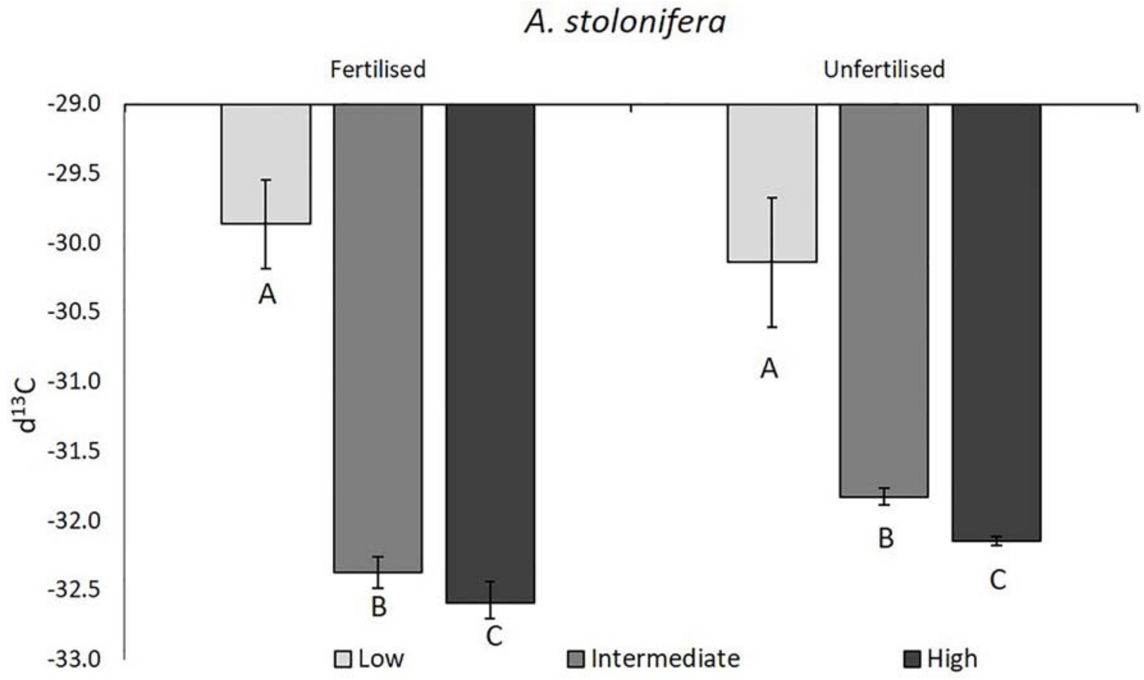

**Figure 3.26. Values for δ$^{13}$C in *A. stolonifera* at three clipping heights, under fertilised and unfertilised conditions. Different capital letters indicate significant differences between clipping heights in fertilised and unfertilised treatments ($p < 0.05$).**

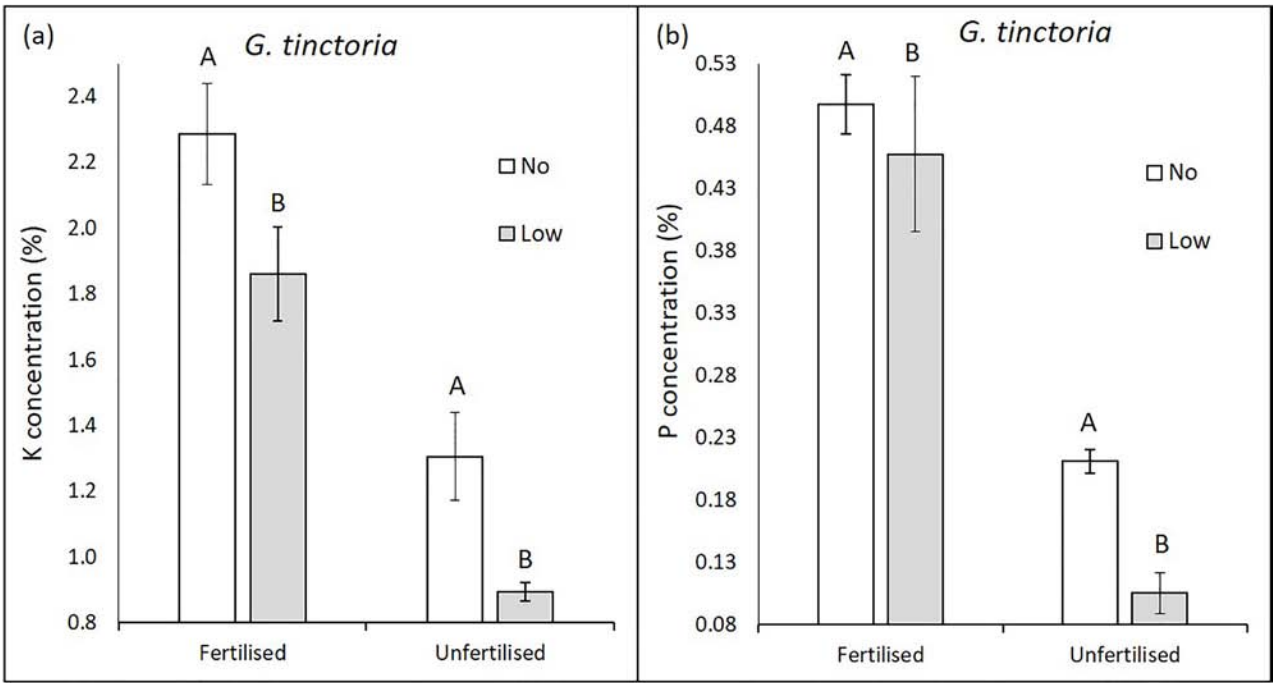

**Figure 3.27. Values for (a) potassium and (b) phosphorus concentrations in *G. tinctoria* tissues grown without competition and under low competition, in fertilised and unfertilised conditions. Different capital letters indicate significant differences between competition levels in fertilised and unfertilised treatments ($p < 0.05$).**

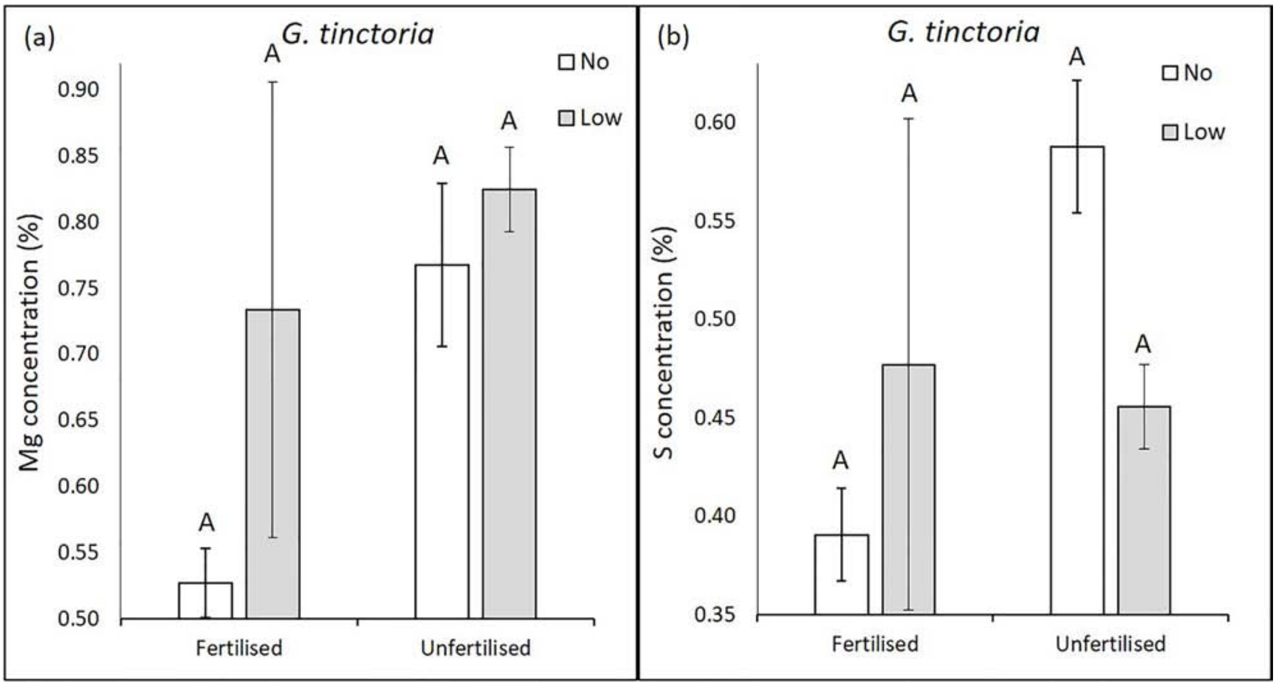

**Figure 3.28. Values for (a) magnesium and (b) sulfur concentration in *G. tinctoria* seedlings grown without competition and under low competition, in the fertilised and unfertilised treatments. Different capital letters indicate significant differences between competition levels in the fertilised and unfertilised treatments ($p < 0.05$).**

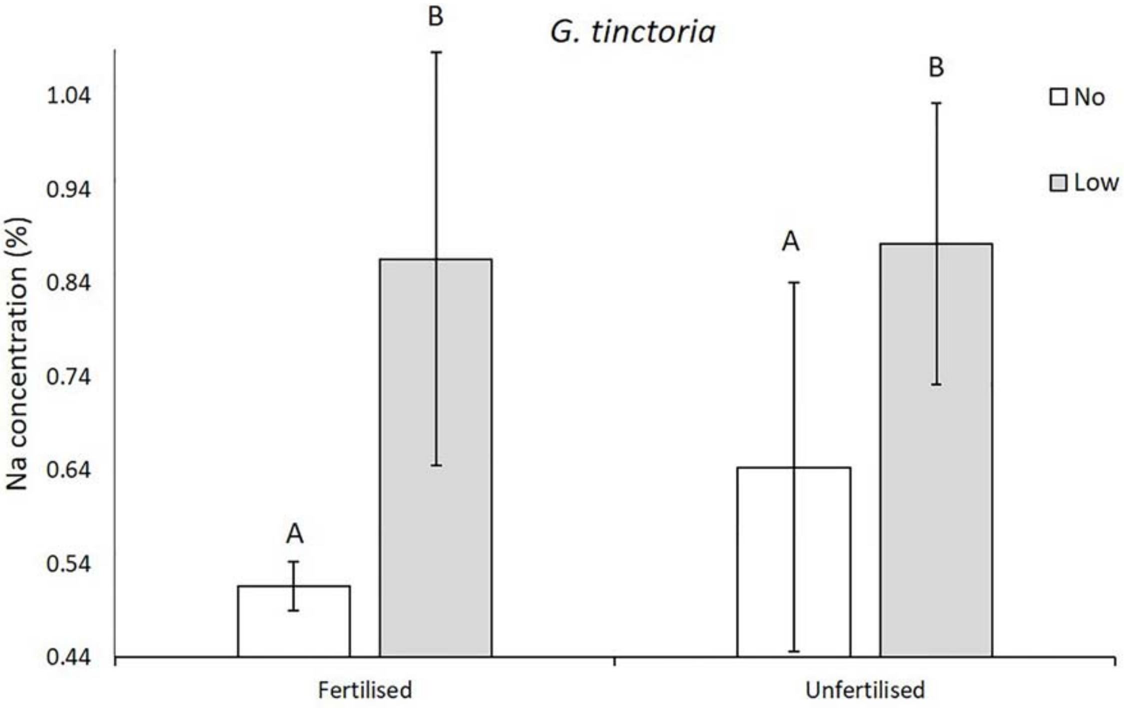

**Figure 3.29. The concentration of sodium in *G. tinctoria* seedlings grown without competition and under low competition, in the fertilised and unfertilised treatments. Different capital letters indicate significant differences between competition levels in the fertilised and unfertilised treatments ($p < 0.05$).**

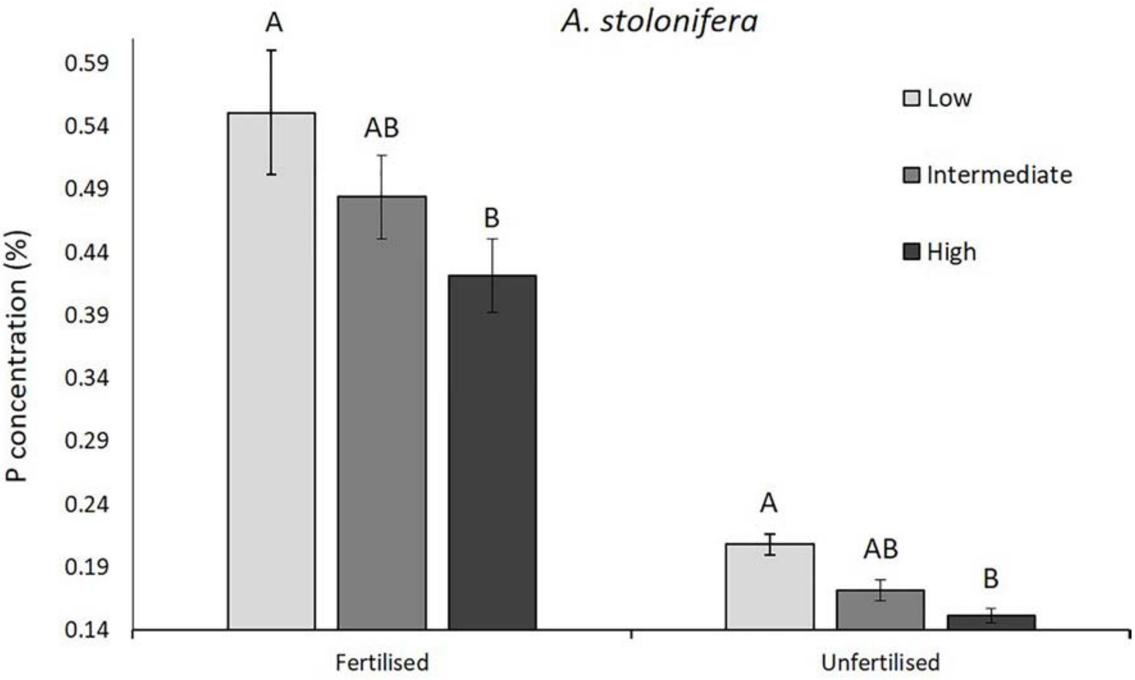

**Figure 3.30. Concentration of phosphorus in *A. stolonifera* seedlings growing in low, intermediate and high clipping treatments, under fertilised and unfertilised conditions. Concentration was significantly increased by competition and significantly decreased by fertilisation, while it was not significantly affected by competition × fertilisation interaction. Different capital letters indicate significant differences between competition levels in fertilised and unfertilised treatments ($p < 0.05$).**

up the extra potassium and phosphorus, but not the extra nitrogen, consistent with the notion that it relies mainly on nitrogen fixation. This result is also supported by the stable nitrogen isotope ratio, which, in the most competitive treatments, gave values consistent with the seedlings depending solely on fixed nitrogen (Figure 3.24). For this reason, fertilisation did not lead to an improved performance of *G. tinctoria* grown in the competition treatments. In contrast, fertilisation generally led to a reduced performance in comparison with unfertilised treatments, expressed as the lower probability of survival (Figure 3.18) and a lower biomass, fewer leaves and a smaller leaf area (Figure 3.19a–c) in comparison with the unfertilised treatments, perhaps because of enhanced grass performance in response to fertilisation. The native *A. stolonifera* was able to take up the extra resources in the fertilised treatments, as nitrogen, phosphorus and potassium were all increased in the grass tissue biomass under fertilised conditions, indicating that, under competitive conditions, native grasses can utilise the additional nutrients to a greater extent than *G. tinctoria* seedlings, when those nutrients are plentiful.

In terms of the mechanism underlying *G. tinctoria's* poor competitive ability, the more negative $\delta^{13}C$ values we found in response to increasing competition suggest that this is related to an effect of resource acquisition on photosynthesis, although this is the opposite of what might be expected if competition for water was the major driver. While there was also little evidence that this was related to tissue nitrogen concentrations, there was a significant difference in tissue phosphorus concentrations between plants with and without competition. While there were also $\delta^{13}C$ isotope-related changes in potassium and nitrogen (positive), these were much smaller than the more than twofold change in tissue phosphorus, suggesting that competition for phosphorus might be the main reason for the differences in plant performance seen in these experiments. Similar results were also found with *A. stolonifera*, suggesting that intraspecific competitive interactions may also be driven by phosphorus availability. Variations in phosphorus uptake could explain the changes in $\delta^{13}C$, as increasing discrimination against the heavier isotope, as competition increases, could be associated with a direct phosphorus limitation of photosynthesis

**Table 3.6. *p*-values calculated from the PERMANOVA, with competition and fertilisation as the main factors**

| | Competition | | Fertilisation | | Competition × fertilisation | |
|---|---|---|---|---|---|---|
| | *G. tinctoria* | *A. stolonifera* | *G. tinctoria* | *A. stolonifera* | *G. tinctoria* | *A. stolonifera* |
| Probability of survival | **0.0013** | | **0.0004** | | **0.0284** | |
| Biomass | | | | | | |
| Total | **0.0001** | **0.0001** | **0.0071** | **0.0001** | **0.0001** | 0.4507 |
| Leaves | **0.0001** | | **0.0064** | | **0.0001** | |
| Petiole | **0.0001** | | **0.0043** | | **0.0001** | |
| Rhizome | **0.0001** | | **0.0183** | | **0.0003** | |
| Root | **0.0001** | | **0.0331** | | **0.0001** | |
| Number of leaves | **0.0001** | | **0.0092** | | **0.0001** | |
| Leaf area | **0.0001** | | **0.0381** | | **0.0004** | |
| *Nostoc*-infected area | 0.467 | | 0.619 | | 0.5183 | |
| C% | | | | | | |
| Leaves | 0.7572 | 0.1317 | 0.8027 | 0.1137 | 0.5611 | 0.1736 |
| Petioles | **0.0117** | | 0.2238 | | 0.524 | |
| Rhizome | 0.1076 | | 0.9363 | | 0.2465 | |
| Roots | 0.658 | | 0.2503 | | 0.2936 | |
| $\delta^{13}C$ | | | | | | |
| Leaves | **0.0001** | **0.0001** | 0.3434 | 0.3149 | 0.0654 | 0.2523 |
| Petioles | **0.0001** | | 0.3106 | | **0.0159** | |
| Rhizome | **0.0001** | | **0.0039** | | 0.1218 | |
| Roots | **0.0015** | | **0.0037** | | 0.0826 | |
| N% | | | | | | |
| Leaves | 0.0892 | 0.2462 | 0.7682 | **0.0001** | 0.5611 | 0.3486 |
| Petioles | 0.1782 | | 0.464 | | 0.7556 | |
| Rhizome | 0.9627 | | 0.8219 | | **0.0297** | |
| Roots | **0.0231** | | 0.2071 | | 0.1052 | |
| $\delta^{15}N_{AIR}$ (leaves) | **0.0223** | 0.7696 | **0.0319** | 0.2167 | **0.0279** | 0.4266 |
| K% | **0.0218** | **0.0279** | **0.0015** | **0.0001** | 0.9161 | **0.0088** |
| P% | **0.0236** | **0.0071** | **0.0011** | **0.0001** | 0.2923 | 0.4864 |
| Mg% | 0.0722 | 0.0823 | **0.0121** | **0.0007** | 0.3147 | **0.0001** |
| S% | 0.8351 | **0.0025** | **0.0411** | **0.0001** | 0.0527 | **0.0003** |
| Ca% | 0.1442 | **0.0006** | 0.3831 | **0.0001** | 0.353 | **0.0001** |
| Na% | **0.0156** | 0.0692 | 0.2653 | **0.0003** | 0.6184 | **0.0154** |
| Fe% | 0.3558 | **0.0003** | 0.4671 | 0.0711 | 0.6877 | 0.2131 |
| Mn% | 0.8146 | **0.0001** | 0.1123 | **0.0001** | 0.3362 | **0.0001** |
| Zn% | 0.9936 | 0.5758 | 0.7131 | **0.0001** | 0.8955 | 0.9995 |
| Co% | 0.914 | 0.6135 | 0.8166 | **0.0002** | 0.6079 | 0.2488 |

**Bold indicates a significant difference ($p < 0.01$).**

(Cernusak *et al*., 2013). It is more difficult to explain the association between low tissue phosphorus levels and $\delta^{15}N$ values. Although the low $\delta^{15}N$ values are consistent with a dependence on nitrogen fixation, phosphorus availability is often considered to limit nitrogen fixation. In studies on legumes subjected to low phosphorus availability, nitrogen fixation may be maintained through internal recycling and the conservative use of phosphorus (Vardien *et al*., 2014). If a similar situation occurs in *G. tinctoria*, this may explain the ability of this species to continue to fix nitrogen despite low external phosphorus availability.

## 3.5 Work Package 5

WP5 combined the results from WP1 (extensively discussed in Gioria *et al*., 2018b, 2019), the experimental results from WP2, WP3 and WP4, and previous knowledge to inform the development of management measures for invasive species in Ireland. Furthermore, account was taken of the 2017 technical note on the management of G. *tinctoria* for the IUCN (Gioria and Osborne, 2017).

The first step, and a priority for every invasive species management plan, should be the prevention of the species' intentional introduction by banning the import, sale, propagation and distribution of plant material, as required under Article 7 of the EU IAS Regulation. This should include online sales, which is a relatively recent trade pathway and can be difficult to monitor and regulate. To prevent the spread of invasive species, attention needs to be given to areas that are in close proximity to the identified source of propagules and it needs to be determined whether or not the environmental conditions are favourable for establishment, growth and reproduction. In susceptible areas, early detection measures ("search and destroy" programmes), repeated annually, would allow both the identification of individual plants and their removal. Removal of *G. tinctoria* should be carried out as soon as possible, as this will be difficult after a few years, and ideally before seedlings can reach sexual maturity or form an extensive rhizome system. However, the implementation of any programme, including any follow-up operations, needs to assess the expected benefits against the expected costs involved. Given the potentially high cost implications, local volunteer groups, if adequately monitored and managed, might have a role to play in the successful management of widespread invasive species. Thus, increasing public awareness of the invasive nature of the species and the best method of control is essential for the prevention of introduction and spread.

Preventing the introduction and spread of invasive species requires an improved knowledge of the ecology and biology of each species, with a view to developing evidence-based long-term and sustainable management measures. To avoid further spread in the future, including post-management recovery and dispersal, additional knowledge of how climate change could affect the distribution and the invasive potential of each species is needed, as is identification of regions/areas that are particularly susceptible to colonisation.

In addition to the general interventions that could limit the introduction and spread of alien plants, species-specific studies are also required for the identification of effective control and eradication programmes. In the case of *G. tinctoria*, short-term trials have shown that chemical or integrated chemical and mechanical measures are effective at reducing the standing biomass, although recruitment through vegetative propagation from the rhizome/rhizome fragments or from the seed bank has been observed at the end of a number of these trials. Longer term monitoring will be required to assess the efficacy of these measures. The present study (WP4) has shown that *G. tinctoria* is not a very strong competitor during the very early stages of its growth, indicating that eradication efforts should focus on the seedling stage, before the formation of a rhizome, within a few days of seedling appearance. After a few years, an extensive rhizome system is usually formed, which would be difficult to remove.

There is a need for new, sustainable and effective control measures that have a minimal environmental footprint. The use of herbicides is likely to face ever-increasing restrictions, while mechanical removal could result in significant soil/land degradation unless it is combined with appropriate restoration measures. It is also evident that any control measures are likely to need to be repeated over several years. Further accounting for the size and demography of established or invasive populations, including the characteristics of the soil seed bank, as well as associated environmental and socio-economic considerations, are also required. The current study has demonstrated very different seed bank strategies between the two invasive species studied (WP2). *I. glandulifera* formed a very small seed bank, which can make eradication easier, while *G. tinctoria* forms a large, long-term, persistent seed bank, which further hampers its control and eradication. Coupled with the different effects on the standing vegetation, with some evidence of differences in their effects on soil resources, any management strategy may need to be tailored to a particular plant invader. It is evident that the impact of *I. glandulifera* on the standing vegetation and the soil seed bank is less than that of *G. tinctoria*, so its removal, in tandem with other appropriate measures, should allow for relatively quick restoration of the original vegetation. However, the extent to which we can generalise these findings to a widespread species such as *I. glandulifera* is uncertain without more information over an extended time frame at a range of sites, given the evidence presented here of site-related effects.

Although often neglected in control and eradication programmes, the soil seed bank plays a key role in the persistence and longevity of an introduced plant species and is an important consideration in the implementation of any restoration measures. More information is required on the probability of successful recruitment from the seed bank after the removal of the standing vegetation and the minimum duration of any control programme required to ensure the successful management of each species.

More attention should be given to alternative, indirect control measures that are focused on the biological, physiological and ecological traits of each species, such as interventions that could target the prokaryotic cyanobacterial symbiont of *G. tinctoria*. The unique and near-obligate intracellular symbiosis involving the nitrogen-fixing cyanobacterium *Nostoc* (Osborne *et al*., 1991; Osborne and Sprent, 2002; see also Gioria and Osborne, 2013) would seem to be an obvious target for control. Biocontrol measures might also be investigated given that this species seems to have few natural enemies in its invaded range, although pathogen infestations have been observed in its native range in Chile (B. Osborne, personal observations). Further consideration should also be given to indirect methods of control that are based on the species' ecology and demography. For instance, the early growth of many invaders, such as *G. tinctoria*, might be counteracted by interventions that stimulate the germination and growth of native species and/or inhibit the germination and growth of the introduced species. The evidence indicates that *G. tinctoria* seedlings germinate and establish late in the year under natural conditions, after the peak in growth of native species, and could be targeted through mechanical removal or mowing/grazing management. Exploitation of any phenological differences will be possible only through a knowledge of the biology and ecology of introduced species, providing a basis for the development of appropriate and environmentally friendly measures to combat the threats posed by alien plant invasions.

# 4 Concluding Remarks

The research carried out for this project set out to evaluate current knowledge on the impacts of two major terrestrial invasive alien plants in Ireland and the strategies used for their management. The study detected a paucity of quantitative information on invasive species impacts on plant communities and ecosystems, with most studies observing decreases in plant diversity in invaded communities. Based on the evidence from this study, a significant investment in long-term research activities is required to protect Ireland's biodiversity and ecosystems, with a focus on how plant invasions influence ecosystem functioning and the provision of essential services, which is likely to be compounded by the added uncertainties associated with global environmental and socio-economic changes. To better understand the wider ecosystem-level impacts of introduced plants, long-term experiments are clearly needed across multiple sites and encompassing different species to monitor not just the invaded communities but also the adjacent uninvaded communities.

Community and ecosystem impacts were clearly shown to be invader dependent (Table 4.1); thus, the importance of the invading species identity for management purposes cannot be overemphasised. While *G. tinctoria* had a negative impact on plant communities, both above and below ground, *I. glandulifera* had an overall neutral effect. Although both the species investigated had significant ecosystem impacts, these differed both qualitatively and quantitatively. For example, *G. tinctoria* invasions were associated with an increase in soil pH but no change in soil phosphorus, whereas *I. glandulifera* invasions were associated with an increase in soil phosphorus but no change in pH. In terms of the effects on the above-ground vegetation, the effects of *G. tinctoria* were quantitatively more significant than those of *I. glandulifera*. Perhaps this is related to *G. tinctoria*'s ability to form large, long-term persistent seed banks, in contrast to *I. glandulifera* (WP2). This is supported by studies from central Europe that found only negligible effects of *I. glandulifera* on above-ground vegetation (Hejda and Pyšek, 2006; Künzi *et al*., 2015; Čuda *et al*., 2017). However, there are still concerns that the species does/may pose risks to semi-natural vegetation communities and plant species, particularly those occurring in low numbers (Prowse, 2001; Hulme and Bremner, 2005; Matthews *et al*., 2015). Furthermore, local environmental factors and management history may also interact with invader identity, indicating that there is a complex matrix of biotic and abiotic interactions that influence ecosystem impacts.

**Table 4.1. Contrasting community and ecosystem effects of the two invaders at the study sites**

| Community effects | *G. tinctoria* | *I. glandulifera* |
|---|---|---|
| Above-ground species richness | Decreased | Decreased |
| Seed bank species richness | Decreased | No effect |
| Ecosystem effects | | |
| Soil available phosphorus | No effect | Increased |
| Soil $NO_3^-$ | Increased | No effect |
| Soil $NH_4$ | Decreased | No effect |
| Soil carbon concentration (%) | Decreased | No effect |
| Soil carbon stocks (C ha$^{-1}$) | Decreased | No effect |
| Soil nitrogen (%) | Decreased | Increased |
| Soil nitrogen stocks (C ha$^{-1}$) | Decreased | Increased |
| Soil microbial biomass | No effect | No effect |
| β-1,4-Glucosidase activity | No effect | Increased |
| Soil pH | Increased | No effect |

The results on soil elemental stocks (WP3) indicated some contrasting effects of the two species (Table 4.1). The increases in phosphorus and nitrogen associated with *I. glandulifera* were, however, small and could be associated with the well-known spatial variability often found in soil elemental stocks. For instance, another study on invasive *G. tinctoria* populations found no significant changes in soil nitrogen and phosphorus stocks (Mantoani, 2019), while small differences were reported in the current study. Consequently, major nutrient enrichment of the catchment and alterations in biodiversity through increased nutrient availability is unlikely. However, it is possible that some enrichment could occur through the loss of dissolved organic or inorganic nitrogen. Measurements of dissolved organic nitrogen concentrations in soils associated with *G. tinctoria* were, however, low, although drainage losses were not measured. At the sites associated with *I. glandulifera* neither dissolved soil organic nitrogen concentrations nor drainage losses were measured. There is, however, some evidence of seasonal increases in soil nitrate concentrations (Mantoani, 2019; Mantoani *et al*., 2020) at the sites invaded by *G. tinctoria*, so this might have resulted in some transient nitrogen enrichment. Further studies would be required to quantify this and its potential significance for both invasions.

It may be somewhat surprising that nitrogen enrichment might not have occurred at the *G. tinctoria*-invaded sites, given that this is a nitrogen-fixing species. We have attributed this largely to the tight control over plant nitrogen, with most accumulated nitrogen being remobilised into the rhizome at the end of the growing season (Mantoani *et al*., 2020), and the lower root-derived rhizodeposition of nitrogen because of a major reduction in root biomass per unit area in invaded areas (Mantoani and Osborne, 2021).

Examination of competitive interactions between *G. tinctoria* and its co-occurring native species has led us to challenge two widespread notions. Firstly, *G. tinctoria* proved to be a poor competitor, as its performance was greatly reduced even under the lowest competitive pressure used, and the probability of survival of young seedlings was also very low. Other traits associated with *G. tinctoria*, such as an ability to reproduce both vegetatively and sexually with high seed production, and a high above-ground biomass productivity, are often cited as features that would confer a competitive advantage. These attributes are, however, a feature of more mature plants associated with the later stage of the invasion process, so early colonisation and establishment is still likely to depend on factors other than a higher competitive ability.

In the fertilised treatments, where *G. tinctoria* was growing alone, an increased nitrogen concentration was found in both above- and below-ground *G. tinctoria* tissues and evidence for the utilisation of an external source of nitrogen based on an increase in $\delta^{15}N$. At face value, these results indicate that *G. tinctoria* might be able to utilise soil mineral nitrogen, when higher concentrations of nitrogen are available, which may challenge earlier findings (Osborne *et al*., 1992) that *G. tinctoria* depends exclusively on nitrogen fixation to meet its nitrogen requirements. If *G. tinctoria* can utilise soil nitrogen when available in excess and in the absence of competition, then it may have an advantage in both nitrogen-depleted environments, through its ability to assimilate atmospheric nitrogen, and nitrogen-enriched environments, through complementary nitrogen uptake.

# 5 Recommendations

This study has attempted to evaluate current management strategies (WP1) and provide management recommendations based on the project results (WP2, WP3 and WP4). Based on the literature, most management practices have focused on the application of herbicide treatments and/or mechanical removal. Given the results of the current study and ongoing environmental concerns, we have stressed the need for alternative site- and invader-specific approaches.

This work has detected a lack of quantitative information on the impacts of terrestrial plant invasions on native communities and ecosystems (WP1). These unquantified impacts might play a critical role in promoting the persistence and spread of primary invaders, i.e. *G. tinctoria* and *I. glandulifera*, and facilitating secondary invasions by other alien plants. There is also a lack of knowledge on the recovery of native species after the removal of an invader, a crucial piece of information for every management and/or restoration measure. This lack of knowledge makes the task of developing control, eradication and restoration measures very difficult. Furthermore, existing management strategies tend to focus on the application of herbicide treatments and/or mechanical removal, which is not always successful or feasible because of the difficulty of accessing invaded sites, and can have adverse environmental impacts (Williams *et al*., 2005). Thus, we have emphasised the need for alternative approaches, such as controlling *G. tinctoria* through its symbiosis by targeting the cyanobacterium *Nostoc* or the use of indirect methods based on an understanding of the demography and ecology of the species (Jones, 2010). The biological control of *I. glandulifera* has been the subject of extensive research, and results from the first UK trials with the rust fungus *Puccinia komarovii* var. *glanduliferae* have shown the potential of this method but have also highlighted some barriers to success (Ellison *et al*., 2020). Further investigations and trials are needed to find new pathogen strains or, through the manipulation of other factors, to make biological control more predictable; they also need to assess the effects of the introduction of the rust fungus on native plant communities (Tanner and Gange, 2020). For both study species, more focus needs to be directed on a prioritised approach that targets geographic areas where they have the greatest impact, such as areas associated with significant losses in biodiversity or which have a major impact on ecosystem services, even though our ability to do this is limited because of a poor understanding of the long-term impacts of plant invasions.

Another pressing issue is that many control measures fail to recognise species- and habitat-specific factors, which, as shown in WP2 and WP3, can be of considerable importance. Thus, we recommend that any control measure should consider the characteristics of each site, including resource availability and the life stage of the invader. Finally, the effective control of existing plant invaders requires an all-island coordinated approach supported by appropriate resources and scientific expertise. Further technical notes should be developed along the lines of those produced for *G. tinctoria* (Gioria and Osborne, 2017) for developing future management plans. This report describes the most cost-effective measures that have been used to control *G. tinctoria*; it also examines novel approaches that could be effective in controlling this species and preventing its spread and accounts for the species' capacity for extensive vegetative propagation and the accumulation of a large, persistent soil seed bank. The detailed report can be found on the CIRCABC website (https://circabc.europa.eu).

The findings in WP2, and specifically the seed bank study, can have important implications for the management of *G. tinctoria* and *I. glandulifera*. Firstly, the study has demonstrated that the two alien plant invaders have substantially different impacts on the plant communities they invade, despite both being listed in the EU IAS Regulation, reinforcing the point that management measures need to be species specific if they are to be effective and economically viable. Invasion by *G. tinctoria* promoted longer term changes that are associated with alterations in both the above-ground and below-ground flora in species-rich grassland communities, and thus restoration of pre-invasion conditions might not be feasible.

In contrast, *I. glandulifera* invasions occurred in species-poor, degraded communities, and have had a minimal overall impact.

Of interest was the finding that the invasive giant hogweed, *H. mantegazzianum,* was not present in areas invaded by *I. glandulifera*, perhaps indicating that *I. glandulifera* displaces it. Given the risks to health associated with giant hogweed, the study findings suggest that its displacement is a beneficial effect of *I. glandulifera* introductions, while its removal is likely to promote colonisation by giant hogweed and other aliens or weedy plants, which were abundant in the immediate vicinity, such as nettle (*U. dioica*).

Although we found that *G. tinctoria* is associated with a long-term persistent seed bank, *I. glandulifera* formed only a very small seed bank, indicating that the management of the seed bank of the former would require extensive efforts and multiple measures repeated over several years. Unfortunately, no reliable information on the time required to successfully exhaust the seed bank of *G. tinctoria* is available from management trials. In contrast, re-establishment of *I. glandulifera* could be managed in a relatively short period of time, by preventing the dispersal of new seeds. From a conservation point of view, the study findings indicate that the removal of both invaders would not be sufficient to promote the establishment of species-rich communities, and that active restoration, involving sowing of species, managing the seed bank of the invaders and manipulating the environmental conditions, would be required to avoid the re-establishment of the invaders and promote the establishment of native communities (Jones, 2010; Tanner and Gange, 2013). Furthermore, the germination trials performed in WP2 revealed a partial tolerance of *G. tinctoria* germination to light, whereas germination of the native co-dominant grass was significantly reduced. This information could mean that the use of traditional management approaches, such as cutting or herbicide treatment, could lead to a reduction in the germination of native grass seeds lying on the soil surface but will not affect the germination of exposed seeds of *G. tinctoria* to the same extent.

Overall, the results from WP3 suggest that the presence of invasive species could have some significant impacts on soil ecosystem functioning; these varied according to both the target variable measured and the species. For example, while the study found a strong positive effect of *G. tinctoria* on soil $NO_3^-$ levels, some other soil parameters and properties were not affected by the presence of this invasive species. Similarly, the study found a positive effect of *I. glandulifera* on soil phosphorus levels, whereas some other soil parameters and properties were not affected by its presence. This study has demonstrated substantially different ecosystem impacts between the two invaders (WP2 and WP3), suggesting that ecosystem-level impacts will ultimately depend on the identity of the invader, its interaction with multiple environmental factors and land use and management history. Thus, this WP has further emphasised the need for species- and site-specific management plans. In terms of restoration, the absence of major effects on soil carbon and nitrogen suggests that restoration with native vegetation might not be impacted by any nutrient-related effects (WP3). However, the presence of a large seed bank could be an issue for the restoration success of areas invaded by *G. tinctoria*, which might take several years to deplete, possibly through regular mowing and/or cutting at the time of seedling recruitment, as discussed previously. In the case of *I. glandulifera*, where there is both little change in soil nutrients and the formation of a small seed bank, restoration should be relatively straightforward.

The results from WP4 could be important for future management plants; however, great care needs to be taken when extrapolating the study results derived from competition experiments under semi-controlled environmental conditions, using only one native species as the main *G. tinctoria* competitor. Furthermore, generalising the study results to a large area or to other plant introductions might also be difficult, as it was found that the site of the invasion and species identity can greatly affect invasion impacts (WP2). However, the results indicate that *G. tinctoria* is a poor competitor, at least in the initial stages of the invasion before a substantial rhizome has been formed.

Based on these findings, any control measures may be more effective if they are applied during the early stages of colonisation or during seedling recruitment, when plants are young and have a high mortality rate, taking advantage of their low probability of survival. As *G. tinctoria* may not be a good competitor, the maintenance of vegetation cover, particularly early/

late in the season, has the potential to supress the survival and early growth of seedlings. Despite this, it is important to stress that *G. tinctoria* establishment can be successful, so it should not be assumed that invasive species that appear to be poor competitors cannot be successful invaders. Clearly, the nature of the recipient community and its characteristics, as well as a range of environmental factors, may determine whether or not a species can establish and become invasive.

The study has shown that nitrogen fertilisation does not improve the competitive ability of *G. tinctoria*, but it does improve the performance of a native grass species. While further work on this is required, fertiliser additions, even at the relatively low concentrations used in this study, could potentially play a role in suppressing the initial stages of *G. tinctoria* invasion. However, the effects of fertiliser additions as part of a management measure on habitats and species in the receiving environment would require careful prior assessments, on a case-by-case basis, and the addition of fertiliser may never be an option for high conservation status sites. As shown in the results from WP4, in cases where *G. tinctoria* invasions are established and where there is low interspecific competition, fertiliser additions could have the opposite effect and increase *G. tinctoria* biomass. This further emphasises that the stage of invasion should be critically assessed and accounted for in any decision-making prior to the implementation of any management interventions.

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# Abbreviations

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|---|---|
| **$\delta^{13}C$** | Ratio of the two stable carbon isotopes, $^{13}C$:$^{12}C$ |
| **$\delta^{15}N$** | Ratio of the two stable nitrogen isotopes, $^{15}N$:$^{14}N$ |
| **BG** | β-1,4-Glucosidase |
| **CIRCABC** | Communication and Information Resource Centre for Administrations, Businesses and Citizens |
| **IAS Regulation** | Regulation on Invasive Alien Species |
| **IUCN** | International Union for Conservation and Nature |
| **PERMANOVA** | Permutational multivariate analysis of variance |
| **UCD** | University College Dublin |
| **WP** | Work package |

**AN GHNÍOMHAIREACHT UM CHAOMHNÚ COMHSHAOIL**
Tá an Ghníomhaireacht um Chaomhnú Comhshaoil (GCC) freagrach as an gcomhshaol a chaomhnú agus a fheabhsú mar shócmhainn luachmhar do mhuintir na hÉireann. Táimid tiomanta do dhaoine agus don chomhshaol a chosaint ó éifeachtaí díobhálacha na radaíochta agus an truaillithe.

## Is féidir obair na Gníomhaireachta a roinnt ina trí phríomhréimse:

**Rialú:** *Déanaimid córais éifeachtacha rialaithe agus comhlíonta comhshaoil a chur i bhfeidhm chun torthaí maithe comhshaoil a sholáthar agus chun díriú orthu siúd nach gcloíonn leis na córais sin.*

**Eolas:** *Soláthraímid sonraí, faisnéis agus measúnú comhshaoil atá ar ardchaighdeán, spriocdhírithe agus tráthúil chun bonn eolais a chur faoin gcinnteoireacht ar gach leibhéal.*

**Tacaíocht:** *Bímid ag saothrú i gcomhar le grúpaí eile chun tacú le comhshaol atá glan, táirgiúil agus cosanta go maith, agus le hiompar a chuirfidh le comhshaol inbhuanaithe.*

## Ár bhFreagrachtaí

### Ceadúnú

Déanaimid na gníomhaíochtaí seo a leanas a rialú ionas nach ndéanann siad dochar do shláinte an phobail ná don chomhshaol:

- saoráidí dramhaíola *(m.sh. láithreáin líonta talún, loisceoirí, stáisiúin aistrithe dramhaíola)*;
- gníomhaíochtaí tionsclaíocha ar scála mór *(m.sh. déantúsaíocht cógaisíochta, déantúsaíocht stroighne, stáisiúin chumhachta)*;
- an diantalmhaíocht *(m.sh. muca, éanlaith)*;
- úsáid shrianta agus scaoileadh rialaithe Orgánach Géinmhodhnaithe *(OGM)*;
- foinsí radaíochta ianúcháin *(m.sh. trealamh x-gha agus radaiteiripe, foinsí tionsclaíocha)*;
- áiseanna móra stórála peitril;
- scardadh dramhuisce;
- gníomhaíochtaí dumpála ar farraige.

### Forfheidhmiú Náisiúnta i leith Cúrsaí Comhshaoil

- Clár náisiúnta iniúchtaí agus cigireachtaí a dhéanamh gach bliain ar shaoráidí a bhfuil ceadúnas ón nGníomhaireacht acu.
- Maoirseacht a dhéanamh ar fhreagrachtaí cosanta comhshaoil na n-údarás áitiúil.
- Caighdeán an uisce óil, arna sholáthar ag soláthraithe uisce phoiblí, a mhaoirsiú.
- Obair le húdaráis áitiúla agus le gníomhaireachtaí eile chun dul i ngleic le coireanna comhshaoil trí chomhordú a dhéanamh ar líonra forfheidhmiúcháin náisiúnta, trí dhíriú ar chiontóirí, agus trí mhaoirsiú a dhéanamh ar leasúchán.
- Cur i bhfeidhm rialachán ar nós na Rialachán um Dhramhthrealamh Leictreach agus Leictreonach (DTLL), um Shrian ar Shubstaintí Guaiseacha agus na Rialachán um rialú ar shubstaintí a ídíonn an ciseal ózóin.
- An dlí a chur orthu siúd a bhriseann dlí an chomhshaoil agus a dhéanann dochar don chomhshaol.

### Bainistíocht Uisce

- Monatóireacht agus tuairisciú a dhéanamh ar cháilíocht aibhneacha, lochanna, uiscí idirchriosacha agus cósta na hÉireann, agus screamhuiscí; leibhéil uisce agus sruthanna aibhneacha a thomhas.
- Comhordú náisiúnta agus maoirsiú a dhéanamh ar an gCreat-Treoir Uisce.
- Monatóireacht agus tuairisciú a dhéanamh ar Cháilíocht an Uisce Snámha.

### Monatóireacht, Anailís agus Tuairisciú ar an gComhshaol

- Monatóireacht a dhéanamh ar cháilíocht an aeir agus Treoir an AE maidir le hAer Glan don Eoraip (CAFÉ) a chur chun feidhme.
- Tuairisciú neamhspleách le cabhrú le cinnteoireacht an rialtais náisiúnta agus na n-údarás áitiúil *(m.sh. tuairisciú tréimhsiúil ar staid Chomhshaol na hÉireann agus Tuarascálacha ar Tháscairí)*.

### Rialú Astaíochtaí na nGás Ceaptha Teasa in Éirinn

- Fardail agus réamh-mheastacháin na hÉireann maidir le gáis cheaptha teasa a ullmhú.
- An Treoir maidir le Trádáil Astaíochtaí a chur chun feidhme i gcomhair breis agus 100 de na táirgeoirí dé-ocsaíde carbóin is mó in Éirinn.

### Taighde agus Forbairt Comhshaoil

- Taighde comhshaoil a chistiú chun brúnna a shainaithint, bonn eolais a chur faoi bheartais, agus réitigh a sholáthar i réimsí na haeráide, an uisce agus na hinbhuanaitheachta.

### Measúnacht Straitéiseach Timpeallachta

- Measúnacht a dhéanamh ar thionchar pleananna agus clár beartaithe ar an gcomhshaol in Éirinn *(m.sh. mórphleananna forbartha)*.

### Cosaint Raideolaíoch

- Monatóireacht a dhéanamh ar leibhéil radaíochta, measúnacht a dhéanamh ar nochtadh mhuintir na hÉireann don radaíocht ianúcháin.
- Cabhrú le pleananna náisiúnta a fhorbairt le haghaidh éigeandálaí ag eascairt as taismí núicléacha.
- Monatóireacht a dhéanamh ar fhorbairtí thar lear a bhaineann le saoráidí núicléacha agus leis an tsábháilteacht raideolaíochta.
- Sainseirbhísí cosanta ar an radaíocht a sholáthar, nó maoirsiú a dhéanamh ar sholáthar na seirbhísí sin.

### Treoir, Faisnéis Inrochtana agus Oideachas

- Comhairle agus treoir a chur ar fáil d'earnáil na tionsclaíochta agus don phobal maidir le hábhair a bhaineann le caomhnú an chomhshaoil agus leis an gcosaint raideolaíoch.
- Faisnéis thráthúil ar an gcomhshaol ar a bhfuil fáil éasca a chur ar fáil chun rannpháirtíocht an phobail a spreagadh sa chinnteoireacht i ndáil leis an gcomhshaol *(m.sh. Timpeall an Tí, léarscáileanna radóin)*.
- Comhairle a chur ar fáil don Rialtas maidir le hábhair a bhaineann leis an tsábháilteacht raideolaíoch agus le cúrsaí práinnfhreagartha.
- Plean Náisiúnta Bainistíochta Dramhaíola Guaisí a fhorbairt chun dramhaíl ghuaiseach a chosc agus a bhainistiú.

### Múscailt Feasachta agus Athrú Iompraíochta

- Feasacht chomhshaoil níos fearr a ghiniúint agus dul i bhfeidhm ar athrú iompraíochta dearfach trí thacú le gnóthais, le pobail agus le teaghlaigh a bheith níos éifeachtúla ar acmhainní.
- Tástáil le haghaidh radóin a chur chun cinn i dtithe agus in ionaid oibre, agus gníomhartha leasúcháin a spreagadh nuair is gá.

### Bainistíocht agus struchtúr na Gníomhaireachta um Chaomhnú Comhshaoil

Tá an ghníomhaíocht á bainistiú ag Bord lánaimseartha, ar a bhfuil Ard-Stiúrthóir agus cúigear Stiúrthóirí. Déantar an obair ar fud cúig cinn d'Oifigí:

- An Oifig um Inmharthanacht Comhshaoil
- An Oifig Forfheidhmithe i leith cúrsaí Comhshaoil
- An Oifig um Fianaise is Measúnú
- Oifig um Chosaint Radaíochta agus Monatóireachta Comhshaoil
- An Oifig Cumarsáide agus Seirbhísí Corparáideacha

Tá Coiste Comhairleach ag an nGníomhaireacht le cabhrú léi. Tá dáréag comhaltaí air agus tagann siad le chéile go rialta le plé a dhéanamh ar ábhair imní agus le comhairle a chur ar an mBord.

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# Managing Invasive Alien Plants in Ireland

Authors: Vasiliki Balogianni, Bruce Osborne, Margherita Gioria and Dario Fornara

## Identifying Pressures

The presence of invasive alien plant species across Ireland and Europe has increased significantly in the past few decades. The impacts of these invasions vary but they can lead to major modifications in ecosystem functioning, including alterations in nutrient cycling and carbon sequestration, habitat loss, species extinctions and decreases in native biodiversity. These impacts substantially affect several economic sectors, such as agriculture, tourism and construction. In Ireland, invasive alien species have caused major changes to native plant communities and ecosystem services, with some threatening not only disturbed habitats but also habitats of high conservation value. This requires further advances in best practice management for preventing the spread of new alien species and managing already invaded ecosystems to account for invader and site-specific effects. Management interventions that recognise the complex effects of plant invaders are becoming increasingly important given the increased global trade in goods and because people, together with global environmental changes, are predicted to promote the expansion of already naturalised populations and the introduction of new alien species.

## Informing Policy

This research project has broadened our understanding of the ecological traits, strategies and impacts of invasive species. This information can be used to refine the way we deal with invasive plants (existing EU/national policy) or influence the types of legislation that might need to be introduced or strengthened (new policy initiatives). Our desktop studies detected a paucity of quantitative information on invasive species impacts on plant communities and ecosystems, highlighting the need for a significant investment in long-term research activities to protect Ireland's biodiversity and ecosystems. Furthermore, our field and experimental studies showed that community and ecosystem impacts were invader dependent; thus, the importance of the invading species identity for management and restoration purposes cannot be overemphasised. Our seedbank studies indicated that restoration of areas invaded by Himalayan balsam might be quite fast and effective, as this species does not form a substantial seedbank and the number of native species in the seedbank were unaffected. In contrast, restoration of sites invaded by Chilean rhubarb may be more problematic and costly and take longer, on account of the loss of native species and the large and persistent seedbank formed by the invader.

## Developing Solutions

The effective control of existing plant invaders requires the development of a coordinated all-island management strategy supported by appropriate resources and scientific expertise, to protect habitats of high conservation value and prevent further loss of biodiversity and ecosystem functioning and services. Given the results of the current study and ongoing environmental concerns, we have stressed the need for alternative site- and invader-specific approaches. Furthermore, based on the findings that Chilean rhubarb is not a very strong competitor during the early stages of establishment, the stage of invasion is an important consideration for management interventions and should be critically assessed and included in decision-making. We suggest that any control measures for Chilean rhubarb may be more effective if they are applied during the early stages of colonisation or during seedling recruitment, when plants are young and have a high mortality, taking advantage of their low probability of survival. We further highlight the importance of the maintenance of native vegetation cover, particularly early/late in the season, which has the potential to supress the survival and early growth of Chilean rhubarb seedlings.

**EPA Research:** McCumiskey House,
Richiew, Clonskeagh, Dublin 14.

**Phone:** 01 268 0100
**Twitter:** @EPAResearchNews
**Email:** research@epa.ie

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