Best-fit relations between hydrological and water quality characteristics of inland wetlands and plant species richness

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SUPERVISORS: DR. IR. EDWIN H. SUTANUDJAJA, NICOLE GYAKOWAH OTOO MSC, DR. MICHELLE T.H. VAN VLIET

Abstract

The increasing global population and climate change have intensified water demands, resulting in increased reliance on groundwater resources. The increasing water demand and diminishing recharge rates pose a significant threat to groundwater levels and discharge. This will lead to alterations in the flow regimes and water volumes impacting wetlands and species richness. Slight modifications in flow regimes and water volumes can result in the loss of specific plant and fish species communities, as these communities are adapted to particular conditions. This research focuses on quantifying relationships between hydrological and water quality variables and plant species richness in wetlands in Australia and England. The approach aims to fill knowledge gaps related to plant species-area relationships and their interactions with hydrology and water quality.

A comprehensive literature review is included about wetlands and identifying which drivers impact the species richness in wetlands. For hydrological datasets, the PCR-GLOBWB model output is used. For the water quality data, the DynQual model output and the World Bank's 'Quality Unknown' data are used. General statistics were calculated and a correlation matrix is used to find the highest correlation with plant species richness. The strength of the hydrological and water quality-species richness relationships are quantified using linear, inverse and power regression. The multiple linear regression is done for all variables including cross-validation.

The results indicate that the wetland area holds the strongest correlation with species richness. Power regression models demonstrate significant prediction, particularly for wetland area and organic pollution (BOD concentrations). The multiple linear regression models show that using one predictor variable is insufficient to explain species richness. The area plays a critical role in improving the fit to species richness. This study can be used to understand the contribution of different driving factors in wetland species richness, for example for effective conservation and sustainable management of wetland ecosystems, especially with the growing anthropogenic pressures taken into account.

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Chapter 1: Introduction

1.1 Problem definition

The growing global population has led to the expansion of water demands increasing the dependency on groundwater resources. A decrease in recharge rates and an increase in water demand can lead to reductions in water levels and discharge (Bierkens & Wada, 2019). This in the long term leads to alterations in flow regimes and water volume, which are major drivers of maintaining wetlands and species richness. Species richness is the number of species in a given area which is used to give an approximate sense of biodiversity in the area. Ecosystem resilience is indicated by biodiversity. An ecosystem with high biodiversity is less susceptible to negative impacts and environmental fluctuations (Elliott et al., 2019).

The ecosystem services and unique biodiversity of rivers, lakes and wetlands will be affected by hydrological alteration. Changes in plant function can be observed when the supply of water is removed for a sufficient length of time (Kløve et al., 2011). This will hinder achieving the UN Sustainable Development Goal 14 and 15 which seek to protect life below water and life on land respectively. Plant species are adapted to specific flow regimes and water volumes and slight alterations may lead to the loss of certain species (Barbarossa et al., 2021; Schipper & Barbarossa, 2022).

A variability in discharge can also affect water quality. Wetlands can act as a filtering and purifying system removing nutrients, pollutants and sediment from the water, improving the water quality. This is under threat due to human activity such as groundwater extraction, land use, pollution, extensive water use for irrigation and extreme weather conditions (*DCCEEW*, 2016b). Therefore it is important to set threshold values to protect aquatic ecosystems and human health but also form economic and recreational points of view. The water quality can also deteriorate due to chemical reactions for example sulphides exposed to oxygen due to groundwater lowering. This leads to the oxidation of sulphides to sulphate and results in the acidification of water, which lowers the water quality (Kløve et al., 2011).

A major threat to biodiversity in wetlands is the alteration of the natural flow regime. The alteration of water regime patterns, as a consequence of changing land uses since settlement, has resulted in shifts in the distribution and abundance of various wetland types and their associated plant communities. Attempts to quantify relationships between flow regulation and biodiversity have, however, been far incomplete or inconclusive, even though wetlands are a major source of biodiversity and highly productive systems. Therefore it is difficult to incorporate impacts of flow regulation on biodiversity in strategic environmental assessments, which makes decision-making more difficult. This hampers the assessment of human impact on wetlands in global biodiversity evaluations (Kuiper et al., 2014).

For understanding spatial patterns of species occurrences and biodiversity conservation, the knowledge of a species' geographic and ecological location is fundamental. Species-area relationships (SAR) are fundamental in comprehending species extension risk and biodiversity patterns. It can enhance the understanding of how species will react and adapt for example to the effects of climate change or other changing physiological characteristics in wetlands (Gap Analysis Project, 2019). Many species-area relationships (SAR) are done for vertebrates such as fish or birds (Elliott et al., 2019; Drakare et al., 2005; Angermeier & Schlosser, 1989). They found that wetland area was the primary driver of species abundance and richness. The species richness tends to increase with increasing sampling area. The variation in abundance and richness could be explained by habitat heterogeneity or wetland characteristics (Elliott et al., 2019). However, there is a gap in knowledge about plant species-area relationships and plant species-water flow and quality relationships and therefore it is necessary to understand how plant species in wetlands are affected and behaving.

This thesis will be focused on quantifying the relationships between hydrological and water quality variables to plant species richness in Australia and England. Australia has a high climate variability which determines the distribution of the wetlands (Brock, 2003). England has a different type of climate than Australia. The distribution of the wetlands and plant species richness are well known and therefore chosen for this thesis.

1.2 Study objective

This research focuses on understanding the best-fit relationships between the physiological characteristics of inland wetlands and plant species richness in Australia and England.

To realize this main objective, two research questions were answered:

1. What are the main drivers impacting species richness in wetlands?
2. What is the relation between wetland site characteristics in particular discharge, wetland area and water quality status such as organic pollution indicated as biological oxygen demand (BOD), nutrients such as total phosphorus (TP) and nitrates/nitrites (NOxN) and salinity (indicated as TDS) in sustaining wetland plant species richness?

1.3 The approach and thesis structure

This study will include a comprehensive literature review about wetlands, plant species richness and drivers impacting wetlands and species richness. A statistical analysis of large-scale data sets on hydrological variables, wetland extent, water quality to plant species richness for different wetlands in Australia and England is included. The statistical analysis includes a matrix correlation plot and regression analyses such as linear, power and inverse regression are used to find the best-fitting relation. After this, multiple linear regression is performed. The strength of the hydrological and water quality-species richness relationship will be quantified using existing models using scaling relationships and data sets from field monitoring. For hydrological datasets, the PCR-GLOBWB model output (Sutanudjaja et al., 2018) is used and for the water quality data, the DynQual model output (Jones et al., 2023) and the World Bank's 'Quality Unknown' data (Damania et al, 2019) is used. The plant species richness dataset is based on the literature studies and datasets. After quantifying the relationship by statistical analysis, the different factors influencing the ecological state can be determined.

This thesis starts (Chapter 2) with a review of wetlands, wetland-dependent species, wetland plants, wetland loss/degradation and their causes, the future of wetlands and 2 case studies. After the literature review, the study area is described for Australia and England (Chapter 3). Then, the methodology is presented (Chapter 4) on how the data is collected, a basic explanation of the used models and how the analysis is performed. After that, the results of England and Australia are simultaneously described (Chapter 5). Then, the results are discussed and some options for future research are provided (Chapter 6). The thesis ends with the conclusions (Chapter 7).

Chapter 2: Literature Review

2.1 Groundwater-dependent ecosystems

Groundwater-dependent ecosystems (GDEs) are ecosystems that are supported by groundwater to maintain their current composition and functioning (Murray et al., 2006). GDEs are a geographically small, yet diverse and valuable part of the biodiversity. Five types of GDEs can be distinguished (Murray et al., 2003). The first ecosystem is terrestrial which shows seasonal or episodic reliance on groundwater. The second system is river base flow, which consists of aquatic and riparian ecosystems in or adjacent to streams/rivers dependent on the input of groundwater base flows. The third is aquifer and cave ecosystems. The fourth are wetlands such as paperbark swamp forests and mound springs. The last are estuarine and near-shore marine ecosystems that use groundwater discharge such as sea-grass beds, salt marches and coastal mangroves (Geoscience Australia, 2023a; Murray et al., 2003). This thesis focuses on inland wetlands.

2.2 Wetlands

The world's landscape contains a variety of permanent and temporary wetlands. which are subdivided into inland wetlands, marine/coastal zone wetlands and human-made wetlands. Wetlands are a critical part of the natural environment and play a key role in supporting biological diversity (DCCEEW, 2016a). About 5–10% of the world's land surface is currently wetland (Kingsford et al., 2016), which is estimated to be 1.2 million square kilometres (Finlayson et al., 2005). Wetlands can be found on every continent for example in deserts to mesic regions, in coastal flats to high mountain ranges and even in Antarctica. More than half of the world's wetlands if found in tropical and subtropical regions (Cronk & Fennessy, 2016). Biodiversity is highly supported in wetlands through wetland productivity, nursery habitats and fresh water. They provide important ecosystem services including food and fibre production, fish, water availability, nutrient cycling, carbon fixation and storage, flood mitigation and water storage; water treatment and purification, culture, education and habitats for biodiversity (Table 1). Wetlands are the most valuable ecosystems and only cover a small part of the world, while they supply more than a quarter of our estimated ecosystem services (Kingsford et al., 2016).

Table 1: Ecosystem services of wetlands (Finlayson et al., 2005).

Services	Examples
Provisioning:	
Food	Fish, fruits and grains
Freshwater	Storage and retention of water for domestic, industrial, and agricultural use
Fiber and fuel	Production of logs, fuelwood, peat, fodder
Biochemical	Extraction of medicines and other materials from biota
Regulating:	
Climate regulation	Source of and sink for greenhouse gases; influence local and regional temperature, precipitation, and other climatic processes
Water regulation	Groundwater recharge/discharge
Water purification and waste treatment	Retention, recovery, and removal of excess nutrients and other pollutants
Erosion regulation	Retention of soils and sediments
Natural hazard regulation	Flood control, storm protection
Supporting:	

Nutrient cycling	Storage, recycling, processing, and acquisition of nutrients
Soil formation	Sediment retention and accumulation of organic matter
Cultural:	
Recreational	Opportunities for recreational activities
Educational	Opportunities for formal and informal education and training

The hydrological cycle is significantly influenced by wetlands due to storage and retention of water, groundwater recharge, low flows, evaporation and floods in wetlands. They also supply water for the population, irrigation, energy and transport. When this is overexploited, it can change the hydrological cycle and thus negatively affect wetlands. Floods can have a positive influence on wetlands, bringing water and nutrient-rich sediment into the ecosystem. This produces fertile soils, habitats and natural irrigation. Reduced flooding can threaten the existence of those wetlands. Wetlands could also contribute to diminishing the destructive nature of flooding. It alters floods by for example reducing the peak or rate, volume and delaying the timing, but when these wetlands are full, they can increase the flood runoff (Acreman & Holden, 2013).

2.3 Wetland-dependent species

Freshwater wetlands contain a relatively large number of species. Terrestrial and marine ecosystems have a larger percentage of known species and the relative species richness in freshwater wetlands is higher (Table 2). The inland wetlands have high levels of endemism due to rivers and lakes acting as physical barriers. A rapid widespread decline is observed in many populations of wetland-dependent species (Finlayson et al., 2005). The Living Planet Index (LPI) measures biodiversity by collecting data and tracking trends in the abundance of large populations of vertebrate species such as amphibians, mammals, birds, fish and reptiles. The global LPI is based on scientific data from 14,152 monitored populations of 3,706 vertebrate species from around the world. The LPI shows a declining trend in the abundance of population. The index showed that freshwater populations have a faster declining rate than other species groups. The freshwater population had an average decline of 81% between 1970 and 2012. Over the same period marine and terrestrial fauna decreased with 36% and 38%. The stronger decline in freshwater species has had more influence on the global decline, which is 58% (Fig.1) (WWF, 2016). Unfortunately, the knowledge about plant species richness trends is not well-researched but is also assumed to decline due to wetland loss and degradation.

Table 2: Habitat extent, species richness and relative Species Richness of Freshwater, Marine, and Terrestrial Ecosystems (Finlayson et al., 2005).

Ecosystems	***Habitat extent (percent of the world)***	***Species Richness (percent of known species)***	***Relative Species Richness***
Freshwater	*0.8*	*2.4*	*3.0*
Marine	*70.8*	*14.7*	*0.2*
Terrestrial	*28.4*	*77.5*	*2.7*

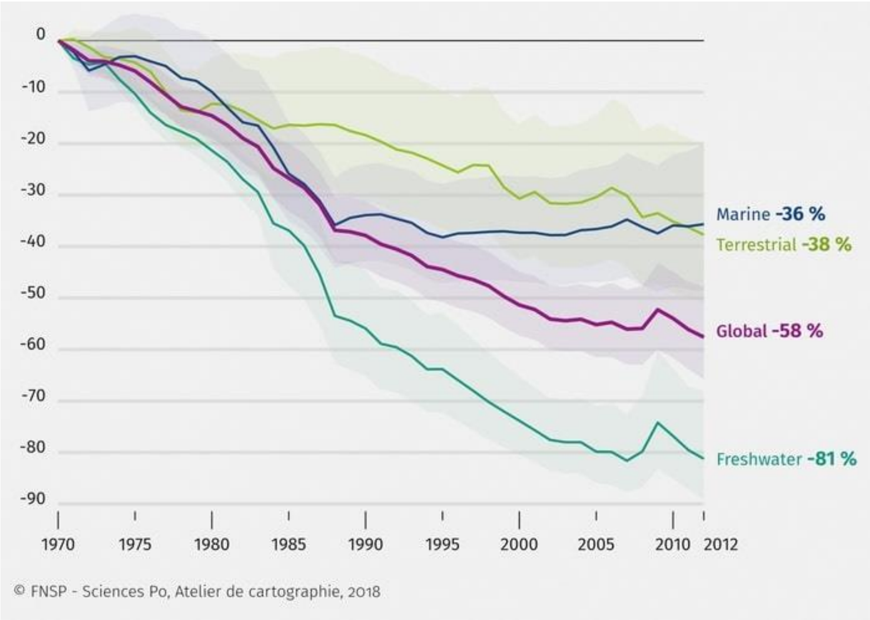


Figure 1: Average decline in abundance of populations by type of habitat (% compared to 1970) (Sciences Po, n.d.).

2.4 Wetland plants

Wetland plants (hydrophytes) are plants growing in water or on a substrate that is at least periodically deficient in oxygen due to excessive water content. Aquatic plants are submerged species or those with floating leaves (Cronk & Fennessy, 2016). Many authors do not make a distinction between wetland and aquatic plants. Many variances between wetland plants are present due to variable hydrology and growth forms. Plants can be emergent, submerged, floating or rooted with floating leaves. The habitat of these plants may extremely vary. There can be permanent or temporary, unpredictable or predictable, saline, freshwater or stillwater habitats (Leck & Brock, 2000).

Wetland plants are categorized based on their growth form, germination and reproduction and are based on how plants grow in physical relationship to the water and soil. The wetland plants are classified into three levels. The first level is divided into terrestrial, amphibious and submerged and is related to the amount of free water in which the species grow. The submerged (10% of species) and terrestrial (32% of species) cannot tolerate submersion or drying and occupy the lower and upper extremes of the wetland zonation. The amphibious species (58%) tolerate and respond to fluctuations in water. this group is further divided into fluctuation-tolerators (38%) and fluctuation-responders (20%) (Brock & Casanova, 1997). Fluctuating water levels cause some amphibious plants to respond by changing growth forms such as leaf shape and length. Other amphibious plants tolerate the variations without changing in form (Leck & Brock, 2000). The amphibious group is further divided into 7 groups that either respond or tolerate for example floating plants or emergent plants (Fig 2).

The amphibious habitat contains less than 5% of oxygen in the water which results in aerobic, hypoxic or anoxic conditions and moderates the daily temperature. The light quality and intensity are affected by depth and materials in the water column. The presence or absence of water in many wetlands cues for germination of plants. Sedimentation, litter deposition patterns, pH, salinity, nutrients and other physical and chemical characteristics are affected by water level fluctuations and this will influence the wetland plants (Leck & Brock, 2000).

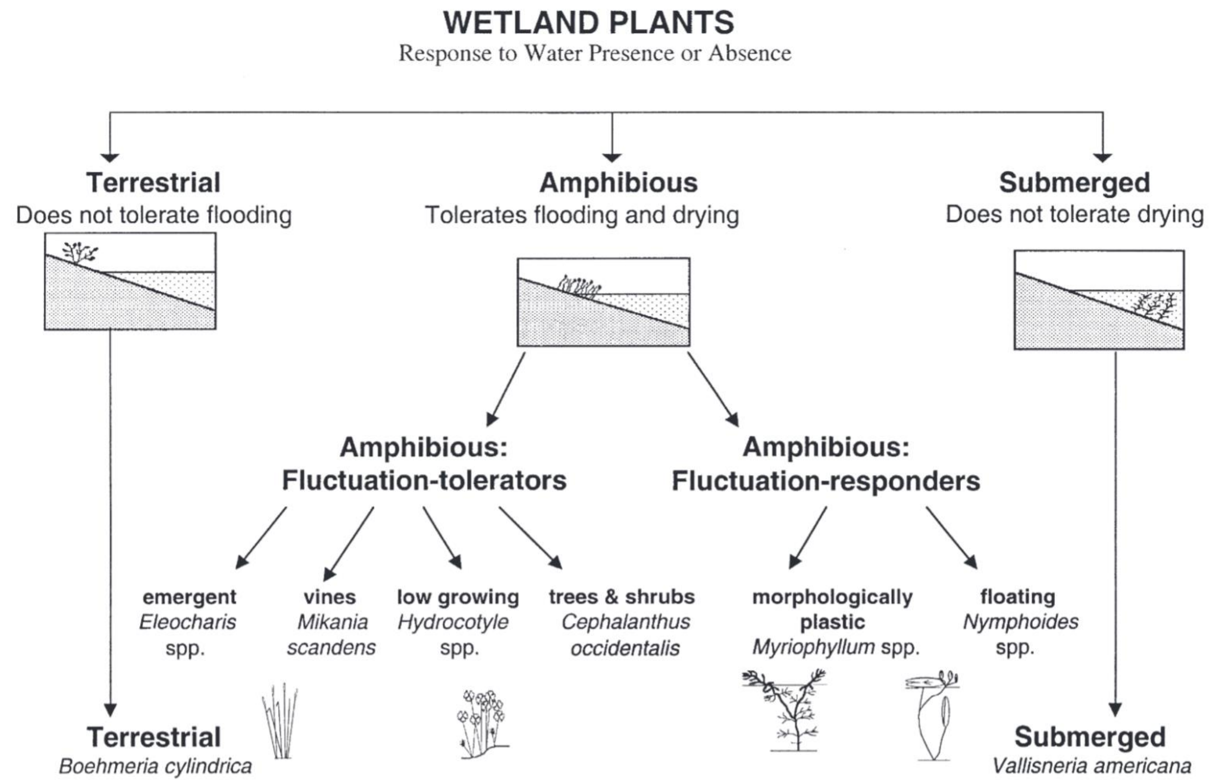


Figure 2: Categorisation of wetland plants (Leck & Brock, 2000).

2.5 Wetland loss

The world has lost a large part about 21% of its wetlands between 1700 and 2020 (Fluet-Chouinard et al., 2023). The losses have been larger and faster for coastal wetlands than inland wetlands. The rapid large-scale conversion of wetlands in Asia promotes a large rate of wetland loss, while wetland loss in Europe and North America has slowed or has remained low since the 1980s. In Africa, the Neotropics and Oceania there is a need to improve the knowledge of change in wetland areas (Davidson, 2014). The national governments established an international convention on wetlands. In 1997, they recognized the Ramsar wetlands, which are representative, rare, unique, or important for conserving biological diversity (DCCEEW, 2022). The convention includes 2185 Ramsar sites covering over 208 million hectares of wetlands (Fig. 3). 168 governmental parties are participating in this convention. It is unclear if this investment in wetlands has influenced the rate of wetland loss. Wetland loss also influences the abundance of species in many parts of the world. Many wetland-dependent species are declining (Davidson, 2014).

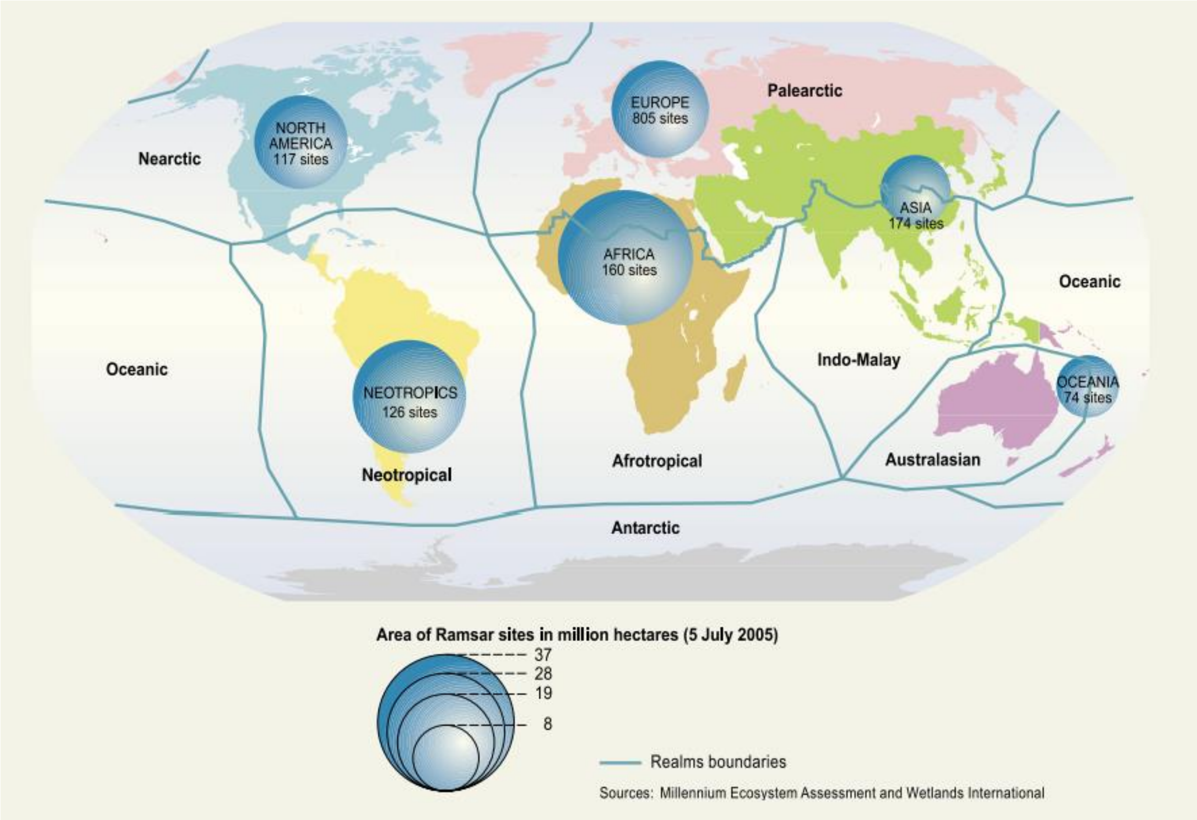


Figure 3: Wetlands listed as internationally important globally (Ramsar Sites) in 2005 (Finlayson et al., 2005).

2.6 Causes of wetland degradation and loss

The direct drivers of degradation and loss of rivers, lakes and wetlands include infrastructure development, land conversion to for example croplands, changes in water use and availability due to for example water withdrawal, pollution, disease control (especially for mosquitoes), the introduction of invasive alien species and overharvesting and overexploitation. The indirect drivers of wetland loss or degradation have been increasing economic development and population growth. These drivers also include the loss of species or reduction of population in these systems (Finlayson et al., 2005).

Draining, infilling and converting wetlands for urban development or agricultural expansion and increased freshwater withdrawal, are the main reasons for wetland degradation (Finlayson et al., 2005). In addition, many wetland habitats are destroyed or vegetation removed in the process of landscaping wetland areas into urban parks (Davis & Froend, 1999). Many species are endangered or lost due to changes in flow regime, modification of inland wetlands, transport of sediments and chemical pollutants and disturbance of migration routes. Since 1960 water behind dams or reservoirs has quadrupled (Finlayson et al., 2005). Dams and diversions cause the water to be more upstream or diverted resulting in reduced water flow, frequency, volume and loss of connectivity to the rivers. Flooding patterns can shift from spring to summer pattern and temperature, channel stability and salinity are affected (Kingsford, 2000).

Agricultural expansion and intensification increased the extensive use of fresh water for irrigation. Wetland clearing for annual crops has resulted in disturbances to aquatic systems, including altered hydrological regimes, sedimentation, changes in thermal and light regimes, the loss of riparian vegetation, nutrient enrichment and salinisation. The replacement of deep-rooted perennial species with shallow cropping species occurred as a consequence of land clearing. Resulting in diminished evapotranspiration, this alteration in the water balance leads to elevated water tables and the mobilization of salt, causing a transition from freshwater to saline conditions. (Finlayson et al., 2011). The expansion of urban and industry caused wetlands to decrease in terms of number and area and the water levels have risen, which increased the surface runoff in the area. However, the increased groundwater extraction decreased the water levels which caused prolonged dry phases. The agricultural and industrial expansion increased the industrial waste (heavy metals) and pollution from pesticides and negatively influenced the species richness (Davis & Froend, 1999).

Groundwater lowering due to for example abstraction for drinking water and irrigation reduces the groundwater recharge, which can reduce groundwater levels significantly and therefore change the water balance in wetlands (Klöve et al., 2014). Candela et al. (2012) researched the effect of the reduction in recharge by reducing groundwater discharge into wetlands in Majorca Spain. Reduced groundwater inflow and sea level rise can negatively affect coastal regions by increasing saltwater intrusion, which negatively influences terrestrial and riparian vegetation. Tujchneider et al (2012) found a loss of biodiversity in GDEs in Santa Fe Argentina after decreasing discharge due to decreasing recharge rates and increasing demand for groundwater. This research supports the negative impact of groundwater lowering in wetlands on plant species richness.

The introduction of invasive plant species is a major cause of local extinction of native plant species. Invasive plants alter the composition of the wetland and habitat structure, lower the biodiversity, modify food webs and change nutrient cycling and productivity. A decrease in native plants causes a decrease in the uptake of nutrients and therefore an increase in nutrients can be observed (Davis & Froend, 1999).

Global climate change will exacerbate the pressure on wetlands which will increase the loss and degradation of many wetlands. It has increased the potential for irreversible change in the ecological state such as loss or decline of species (Finlayson et al., 2011). Climate change is predicted to lead to an increase in precipitation over more than half of the earth's surface. This will lead to more water available to ecosystems and society. However, this precipitation pattern will not be universally distributed over the earth (Finlayson et al., 2005). Some areas will experience less precipitation and more droughts which results in more pressure on for example water demand. Climate change will increase the sea level, storm and tidal surges and change storm intensity and frequency and this will change the river flow and sediment transport (Flood, n.d.). This is a high threat to coastal wetlands. Higher temperatures due to climate change heighten the risk of increased evaporation and alter the flow regime. Vector-borne diseases such as malaria and waterborne diseases such as cholera are increasing due to climate change and expanding urbanization (Finlayson et al., 2005). An increasing number of wetlands are therefore treated with Abate which negatively influences species richness. The projected climate change has harmful impacts on many wetlands despite the benefits of increased precipitation. It's more difficult for wetland species to relocate or adapt to these circumstances which will result in loss of species (Davis & Froend, 1999).

2.7 Case studies

This case study presents the effect of groundwater salinity and surface water salinity on plant species richness in coastal pine forests and some wetlands north of the Bevano River. It shows a decrease in plant species richness with increasing salinity. In a part of the study area, the plant species richness increases due to an increase in salt-tolerant plants (halophytes). The species richness is not only dependent on water salinity but also on microclimate, soil properties, atmospheric conditions, pollution, etc. However, due to the relatively small size of the area, it is concluded that salt tolerance of vegetation is a major factor controlling species richness. They concluded a dramatic decrease in species richness when the water salinity exceeds the 3 g/l threshold (Fig. 4) (Antonellini & Mollema, 2010).

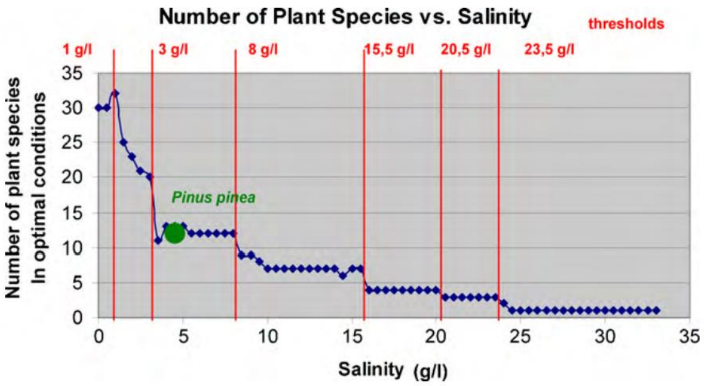


Figure 4: The graph shows the number of plant species out of a total of thirty-nine, typical of the study area that can tolerate groundwater containing the salinity at the x-axis. The salinity tolerance thresholds for plant species are 1, 3, 8, 15.5, 20.5, and 23.5 g/l (Antonellini & Mollema, 2010).

Another case study in Pre-Alps of Switzerland examines the effect of habitat fragmentation, which leads to a decrease in habitat area and an increase in isolation of habitat islands, on species richness. The relation between wetland area and species richness is positive using the power law and concluded that large habitat islands contain fewer species than sets of smaller habitats with the same total area (Fig. 5) (Peintinger et al., 2003).

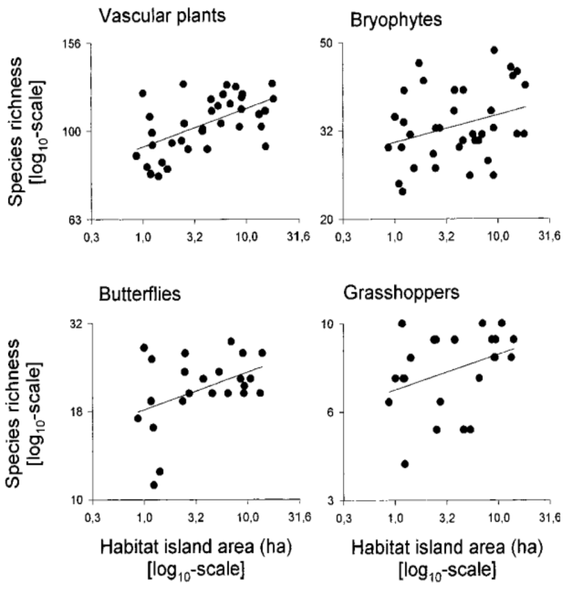


Figure 5: The species-area relationship for four taxonomic groups (Peintinger et al., 2003).

2.8 The future of wetlands

The MA (Millennium Ecosystem Assessment) developed four scenarios up to the year 2050 to examine possible futures for the wetlands and their impact on human well-being. The scenarios explored two different approaches for ecosystem management (reactive and proactive) and explored two global development paths. Proactive management attempts to maintain the ecosystem services for the long term, while reactive management addresses problems after they become obvious (Adeel et al., 2005).

- *Global orchestration:* This scenario depicts a globally connected society with reactive ecosystem management. It takes steps to reduce poverty and inequality and increase economic growth and public goods such as infrastructure and education. In this scenario, economic growth is the highest but it has the lowest population in 2050.
- *Order from Strength:* This scenario represents a regionalized world with reactive ecosystem management. The world is fragmented with emphasis on security, protection and little attention to public goods. Economic growth rates are the lowest and population growth the highest.
- *Adapting Mosaic:* This scenario represents a regionalized world with proactive ecosystem management. Regional watershed-scale ecosystems are the focus and local ecosystem management strategies are common. Economic growth is low but increases with time and the population is nearly as high as in Order from Strength.
- *TechnoGarden:* This scenario is globally connected with proactive ecosystem management. The scenario relies strongly on green technologies. Economic growth is relatively high and accelerates, while the population in 2050 is in the midrange (Adeel et al., 2005; Finlayson et al., 2005).

In the Global Orchestration and Order from Strength scenarios, the area of wetlands is expected to decrease while the TechnoGarden and Adapting Mosaic remain relatively unchanged. The wetland area decreases because of the expansion of agricultural land which is in the Order of Strength the largest. The Adapting Mosaic has a large agricultural expansion but due to proactive approaches, the wetland loss is mitigated. TechnoGarden has the smallest pressure on the environment. In all four scenarios, habitat loss is projected to lead to a decline in the local diversity of species and ecosystem services (Finlayson et al., 2005).

Toward 2050 climate change have a pronounced effect, especially in the Global Orchestration, Order from Strength, and Adapting Mosaic scenarios. The demand for provisioning services such as water and food increases in all four scenarios due to growth in economies and population. This will lead to further stresses on the ecosystem (Alcamo et al., 2005). Massive increases in water withdrawal are expected under the Global Orchestration and Order from Strength scenario, causing deterioration of the water quality (Finlayson et al., 2005). The combination of increasing water withdrawal, deterioration of quality and declining water availability during parts of the year leads to increases in water stress in large parts of the world (Van Vliet et al., 2021). The water availability under these MA scenarios is projected to diminish by 30% of the world's rivers. A higher deterioration of services is expected in the scenarios with reactive approaches while a lesser decline is expected in proactive management (Finlayson et al., 2005).

2.9 Summary

Wetlands are vital ecosystems that support a diverse range of plant and animal species while providing numerous ecosystem services. Wetlands consist of various types, such as inland wetlands, coastal/marine wetlands, and human-made wetlands which cover about 5-10% of the Earth's land surface. They play essential roles in biodiversity conservation, water purification, flood control, and climate regulation.

Wetlands face significant threats from human activities such as urbanization, agriculture, and infrastructure development, as well as pollution and invasive species introductions. Alterations in hydrological regimes, disrupt wetland ecosystems' natural functions, leading to declines in water quality, altered flood patterns, and habitat loss. These pressures have led to the loss and degradation of wetlands worldwide, impacting the species richness and reducing the provision of ecosystem services. Global climate change exacerbate the pressure on wetlands. It has increased the potential for irreversible change in the ecological state.

Several case studies illustrate the adverse effects of these threats on wetland ecosystems. For example, groundwater salinity and habitat fragmentation/ decreasing habitat have been shown to decrease plant species richness in wetlands, highlighting the vulnerability of these ecosystems to environmental changes.

Looking ahead, proactive ecosystem management approaches are crucial for mitigating wetland loss and species richness decline. Scenarios for the future of wetlands emphasize the importance of sustainable management practices to conserve these valuable ecosystems and ensure their resilience in the face of ongoing environmental challenges. Protecting and restoring wetlands is essential not only for preserving biodiversity but also for safeguarding the ecosystem services they provide, which are vital for human societies worldwide.

Chapter 3: Study area

3.1 Australia

Climatic patterns particularly rainfall, evaporation and morphological and topographic features largely determine the distribution of wetlands in Australia. The southern coastal area of Australia is characterized by a predictable winter-wet temperate climate, the north of Australia has a predictable summer-wet climate and the middle of Australia (two-thirds of Australia) has a large arid and semi-arid climate and is characterized by unpredictable rainfall patterns. Permanent and temporary wetlands occur often near the coast. Permanent wetlands occur where rainfall exceeds 1000 mm per year, this can be seen in Fig. 6 as the shaded area. In the semi-arid and arid centre saline and episodic wetlands dominate (Brock, 2003). Water levels in wetlands are usually at a maximum in spring (September and October) and a minimum in autumn (March and April) (Davis & Froend, 1999).

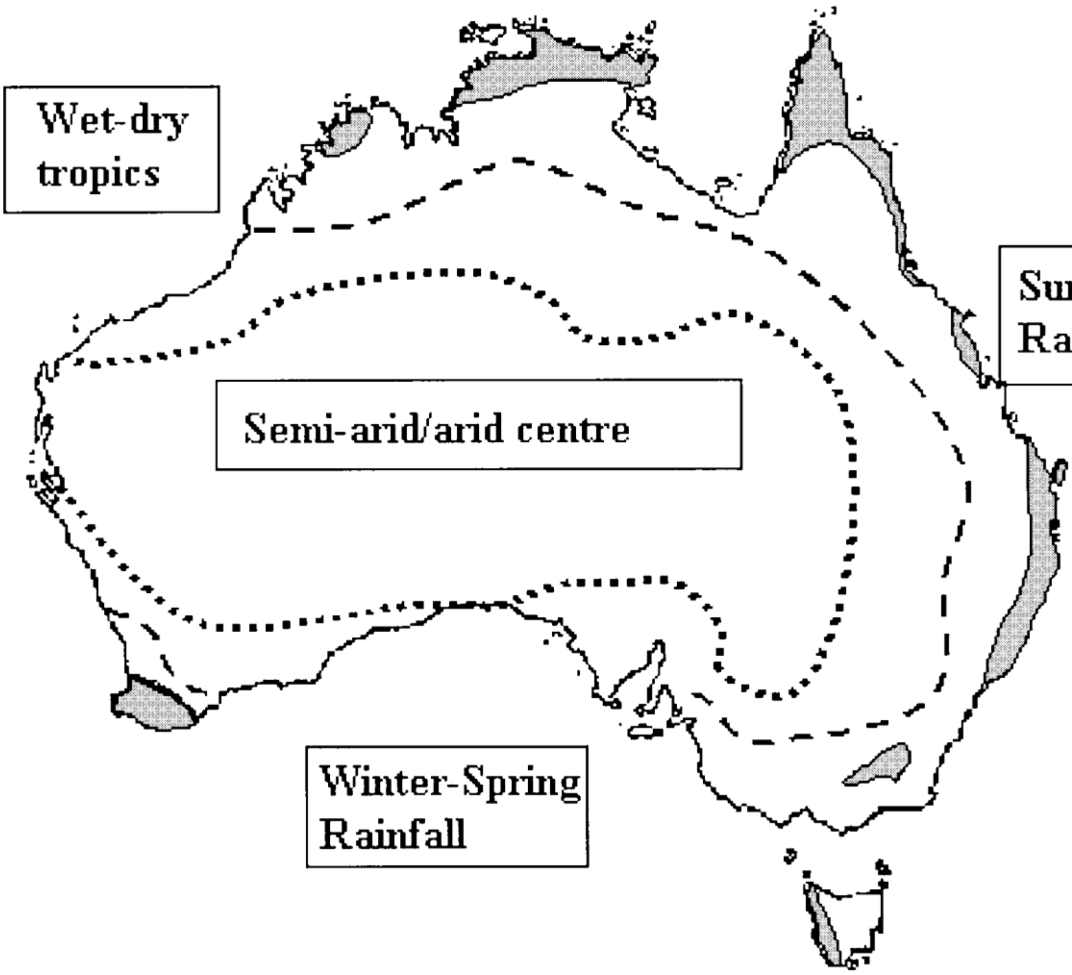


Figure 6: Climatic patterns of Australia. Contours of annual rainfall dotted line = 250 mm, broken line = 500 mm, shaded = >1000 mm (Brock, 2003).

Mapping the distribution of wetlands in Australia is undertaken by Geoscience Australia (Table 3). About 23,007,700 ha of natural wetland is mapped with an additional 66,700 ha of man-made wetlands. Four river basins consist of the largest wetland area (87%), which are the Lake Eyre Basin (31%), the Murray-Darling Basin (23%), the Timor Sea (18%) and Gulf of Carpentaria (15%) (Finlayson et al., 2011). Some wetlands are representative, unique, rare or important for conserving biological diversity, these wetlands are called Ramsar wetlands. Currently, 67 Ramsar wetlands are present that cover 8.3 million ha (Fig. 7) (DCCEEW, 2022).

Table 3: Locations and area (ha) of major wetland types in Australia, based on GeoScience Australia (Finlayson et al., 2011).

Basin	Total natural wetlands[a]	Lakes	Floodplains[b]	Swamps	Marine wetlands[c]	Rivers[d]	Man made wetlands[e]
1. North East Coast	9,813.80 (4)	469.76	3,224.65	1,140.37	3,221.78	1,757.24	1,132.93
2. South East Coast	7,222.36 (3)	2,588.56	2,438.47	870.64	842.63	482.06	696.95
3. Tasmania	1,724.83 (1)	364.11	100.62	540.60	480.97	238.53	1,199.60
4. Murray-Darling Basin	53,416.56 (23)	10,066.29	39,500.09	2,985.75	16.07	848.38	1,996.23
5. South Australian Gulf	7,542.77 (3)	5,860.36	465.57	13.79	1,177.67	25.38	98.69
6. South West Coast	11,965.06 (5)	6,141.72	5,138.29	437.62	79.81	167.61	115.26
7. Indian Ocean	17,089.32 (7)	3,693.70	4,824.89	97.87	3493.25	4,979.60	115.27
8. Timor Sea	42,406.43 (18)	737.51	23,526.56	2,669.56	11,925.26	3,547.54	1,101.17
9. Gulf of Carpentaria	34,838.10 (15)	648.44	22,256.51	1,725.69	7,362.34	2,845.11	128.10
10. Lake Eyre Basin	72,212.60 (31)	25,292.04	43,686.57	1,334.01	0.00	1,899.98	18.90
11. Bulloo-Bancannia	9,436.04 (4)	607.58	7,696.99	1,121.02	0.00	10.45	8.49
12. Western Plateau	53,576.15 (23)	44,116.12	7,856.14	241.39	448.09	914.41	46.22
Total	23,3077.84	75,095.41	109,847.67	7,189.54	23,228.94	17,716.29	6,657.79
% of types	n.a.	31.3	45.8	3.0	9.7	7.4	2.8

Anthropogenic changes to wetlands, occurring at scales ranging from individual wetlands to catchments and landscape scales can alter the availability of plant habitats and consequently impact the diversity of wetland plants (DCCEEW, 2016a). Australia is a groundwater hotspot. Groundwater is 17% of the accessible water resources and accounts for over 30% of the total water consumption. The groundwater use exceeds the rate at which groundwater is recharged, which can negatively influence species richness (Geoscience Australia, 2023b). Wetland loss and degradation in Australia have been severe and are still occurring, influencing the plant species richness. Currently, a decrease in plant species is observed in Australia (Fig. 8) (SoE, 2021). Especially in areas with wetlands (south of Western Australia, south of South Australia, Victoria and at the coast of Queensland and New South Wales), a decrease is observed.

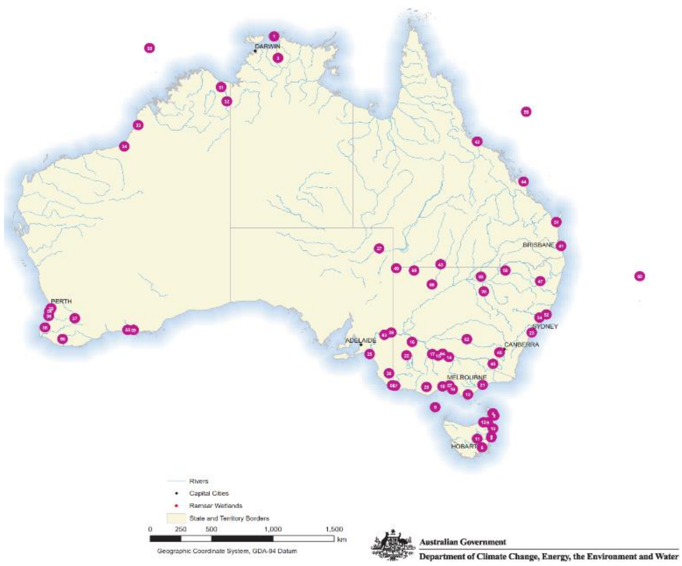


Figure 7: Locations of the Ramsar wetland sites (Australian Ramsar Wetlands - DCCEEW, 2022).

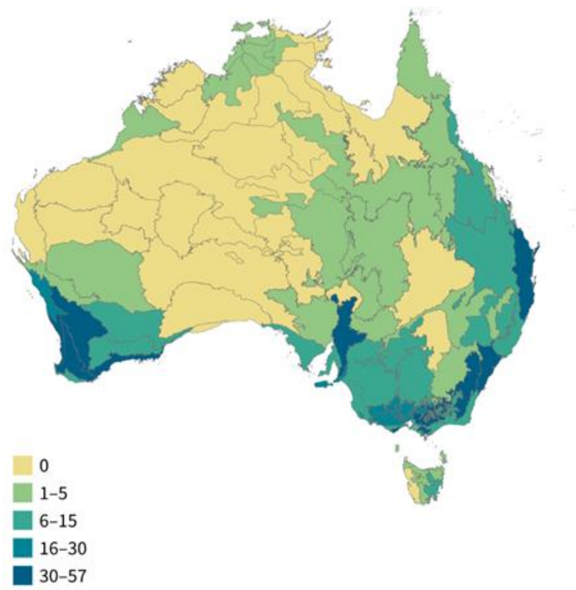


Figure 8: Number of declining plant species richness per bioregion (SoE, 2021).

3.2 England

Wetlands in England are an important part of the landscape and cover almost 10% of the terrestrial land area (Fig. 19a). These wetlands include habitats such as fens, bogs, marshes, wet grassland, mudflats and floodplains (Dawson et al., 2003). The first Ramsar site was designated in 1976 and England consists now of 71 wetlands with international importance.

England has compared to other countries a relatively large number of Ramsar sites, although they are smaller in size (JNCC, 2022).

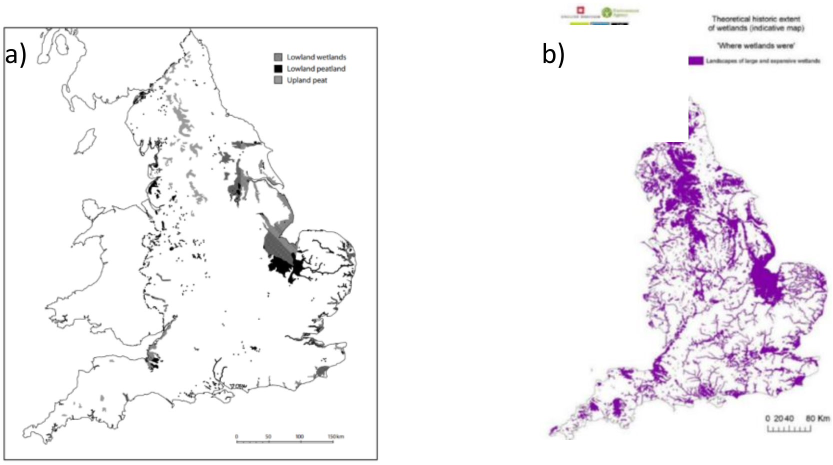


Figure 9: a) Distribution of England's wetlands (Olivier, 2004) and b) a map of the historic extent (Stratford & Acreman, 2016).

The wetland extent and quality of the wetlands have decreased dramatically in England (Fig. 9a/b), with the highest loss occurring during the period after the Industrial Revolution (1760 to 1840). Only 10% of the original wetland area remains in the UK. The degradation and loss of wetlands in England are due to erosion, drainage, land use change, water abstraction, peat extraction and other impacts of modern life such as urban and industrial development. About 40% of the wetlands are currently under productive agriculture, producing 70% of the UK's food (Stratford & Acreman, 2016). The Monuments at Risk in England's Wetlands project estimated the presence of 18000 wetland sites but almost 3000 have been destroyed and 10000 have suffered damage or destruction. Only 5000 sites are likely to survive (Olivier, 2004).

In contrast to Australia, England is much smaller and climate is not the main factor in how wetlands are distributed over the country. The climate is oceanic, with quite cold, rainy winters and very mild or warm summers. The groundwater levels are highly variable and long-term trends may be influenced by different factors such as land-use change, land cover change and abstraction change (Bloomfield et al., n.d.). The wetland plant species richness in England is not well researched. The Countryside Survey made an index of the number of different plant species in different habitats between 1990 and 2007 (Fig. 10). It shows and increase in plant species richness in arable and horticultural land in both long-term (62%) and short-term (26%). No change was observed in improved grassland, broadleaf woodland and ground flora

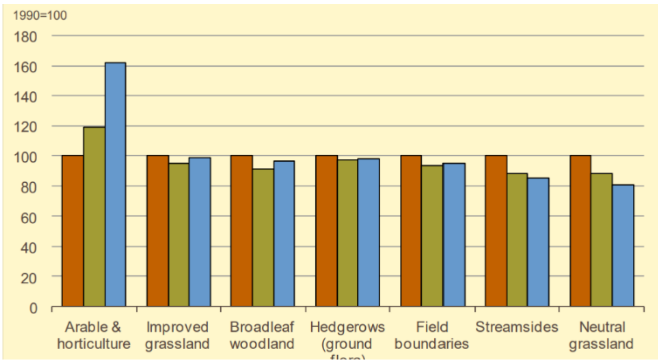


Figure 10: The index of plant species richness for England from 1990 to 2007 (DE7 Index of Plant Species Richness: England, 2007).

hedgerow. A decline in plant species richness in field boundaries (5%), stream sides (15%) and neutral grassland habitats (19%) between 1990 and 2007 (DE7 Index of Plant Species Richness: England, 2007)

Chapter 4: Methodology

4.1 Data

The area (km2) and location of 479 wetlands in Australia (Fig. 11a) were taken from A Directory of Important Wetlands (Table 4) (Directory of Important Wetlands, n.d.). These important wetlands were used in the Atlas of Living Australia where the species richness per location could be taken in a timeframe between 1850 and 2023 (Belbin et al., 2021). For England 71 wetlands were taken (Fig. 11b). The shapefile, location and area were taken from Ramsar Wetland UK (Defra group ArcGIS Online organisation, 2017). The shapefile was used in the spatial analysis portal of the NBN Atlas to acquire plant species richness per wetland in a timeframe between 1850 and 2023 (NBN, n.d.). Different hydrological variables were chosen, because flow alterations (e.g. change in discharge and groundwater recharge) have a large impact on wetlands and flow regulation is an important ecosystem service. When these alterations happen it will affect the water quality. So it is important to also include variables that can change the water quality. For example, agricultural practice can increase phosphor, nitrate and nitrite load. This will change the organic content in the surface water and therefore a change in biological oxygen demand. Changing groundwater depth affects the fresh/salt water distribution. All these variables have some influence on wetlands and therefore need to be taken into account.

The variables that were used in the analyses are discharge (m3/s), groundwater recharge (m/day) and evaporation (m/year) from the PCR-GLOBWB model (Sutanudjaja et al., 2018). The groundwater depth (m) from the GLOBGM model (Verkaik et al., 2024) and the salinity (TDS) (mg/L) and Biological Oxygen Demand (BOD) (mg/L) were taken from the DynQual model (Jones et al, 2023). The total phosphorus (TP) and nitrate/nitrite (NOxN) (mg/L) and the electrical conductivity (mS/cm2) are taken from World Bank's 'Quality Unknown' data (Damania et al, 2019). The variables used in PCR-GLOBWB, DYNQUAL and Quality Unknown are taken in a timeframe between 1992 and 2010. The BOD was also included in the Quality Unknown data to compare it with the BOD from the PCR-GLOBWB model. The Quality Unknown data relies on machine-learning algorithms known as Random Forests. It seeks to find the combination of factors that explain observed water quality by estimating thousands of decision trees (Desbureaux et al, 2022).

Table 4: Used datasets per area.

Area	Data/Model	Source	Resolution
Australia	Species richness	Atlas of Living Australia – Open access to Australia's biodiversity data (ala.org.au)	points
Australia	Wetland area and location	Australian Wetlands Database - Directory Wetland Search (environment.gov.au)	coordinates
England	Species richness	https://nbnatlas.org/	points
England	Wetland area and location	Ramsar (England) | Ramsar (England) | Natural England Open Data Geoportal (arcgis.com)	Polygons
World	Water quality data	Quality Unknown (wbwaterdata.org)	Mean of 3 x3 0.5° grid cells (~50 km)
World	PCR-GLOBWB	GMD - PCR-GLOBWB 2: a 5 arcmin global hydrological and water resources model (copernicus.org) (Sutanudjaja et al.,2018)	5 arcmin (10 x ~10 km)
World	GLOBGM (groundwater depth)	GMD - GLOBGM v1.0: a parallel implementation of a 30 arcsec PCR-GLOBWB-MODFLOW global-scale groundwater model (copernicus.org) (Verkaik et al.,2024)	30arcsec (~1 km)
World	DynQual	GMD - DynQual v1.0: a high-resolution global surface water quality model (copernicus.org) (Jones et al. 2023)	5 x 5 arcmin (10 x ~10 km)

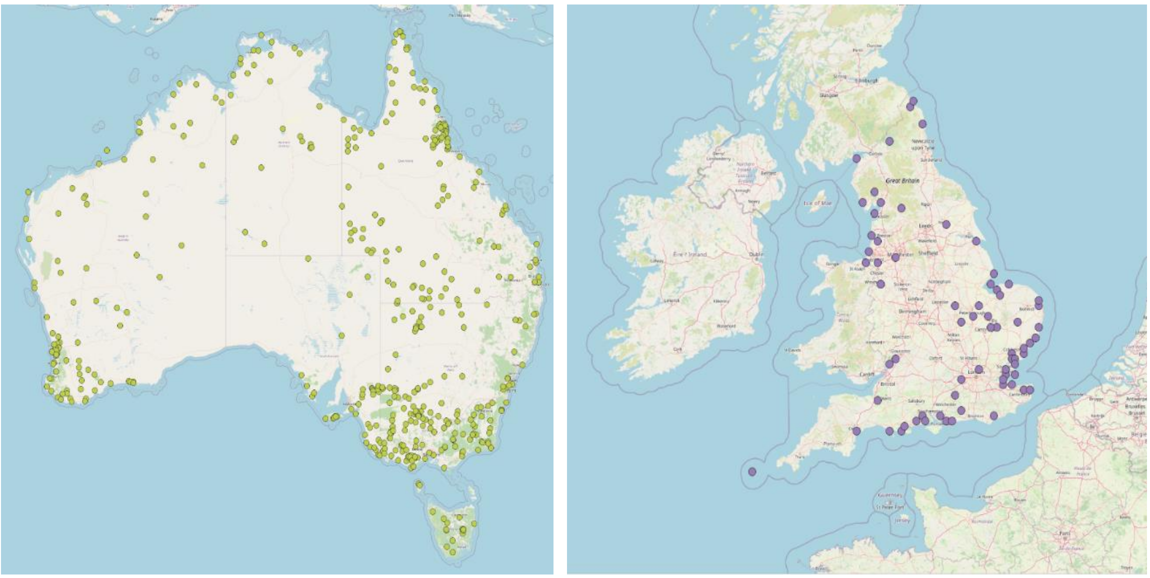


Figure 11: Location of the used wetlands a) Australia and b) England.

PCR-GLOBWB hydrological model

The PCR-GLOBWB model (Sutanudjaja et al., 2018) is a grid-based global hydrology and water resources model developed at Utrecht University. The model can predict various state variables such as stream flow (Table 5), it considers also human water use. The model consists of three layers (Fig.12). S1 and S2 are two soil moisture storages in the upper and lower soil layer and S3 is the groundwater storage. Each of those layers has its own water flow and water exchange between them. The groundwater Storage has Qbf (baseflow) and Inf (infiltration from riverbed to groundwater). The layers S1 and S2 have Qdr (direct surface runoff, generated by snowmelt and rainfall) and Qsf (interflow or stormflow). All these surface fluxes combined are Qchannel (channel flow). Precip (precipitation) and Evap (evapotranspiration) interact with the top layer and atmosphere. Human interaction through livestock, industry, domestic use and irrigation are also taken into account. The red dashed arrow lines are return flows from surface water use. The blue dashed arrow lines are return flows from groundwater use. The thin solid blue line indicates groundwater abstraction and the red solid line indicates surface water withdrawal (Shen et al., 2022). The PCR-GLOBWB model has a 5 arc-minute resolution ((~10 km at the equator). The internal time stepping for hydrodynamic river routing varies while the time steps for water use and hydrology is one day. For each time step the human water use is integrated. First, the water demand for irrigation, livestock, households and industry is estimated. Then the demands are translated into actual withdrawal from the surface water and groundwater and last the return flows and consumptive water use are calculated per sector (PCR-GLOBWB 2.0, n.d.).

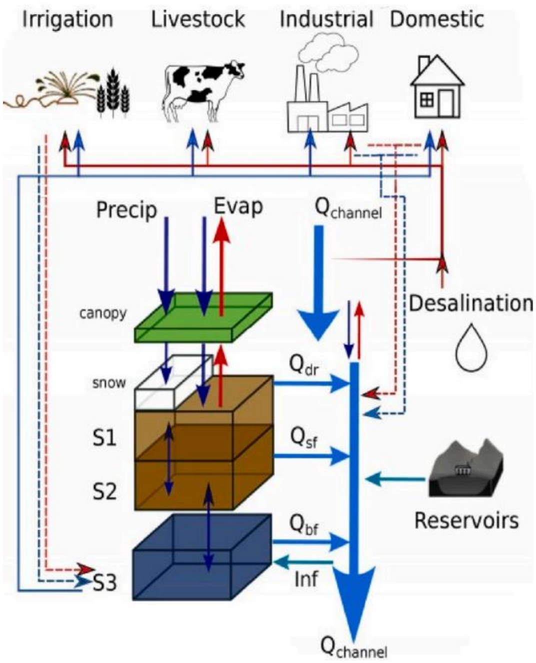


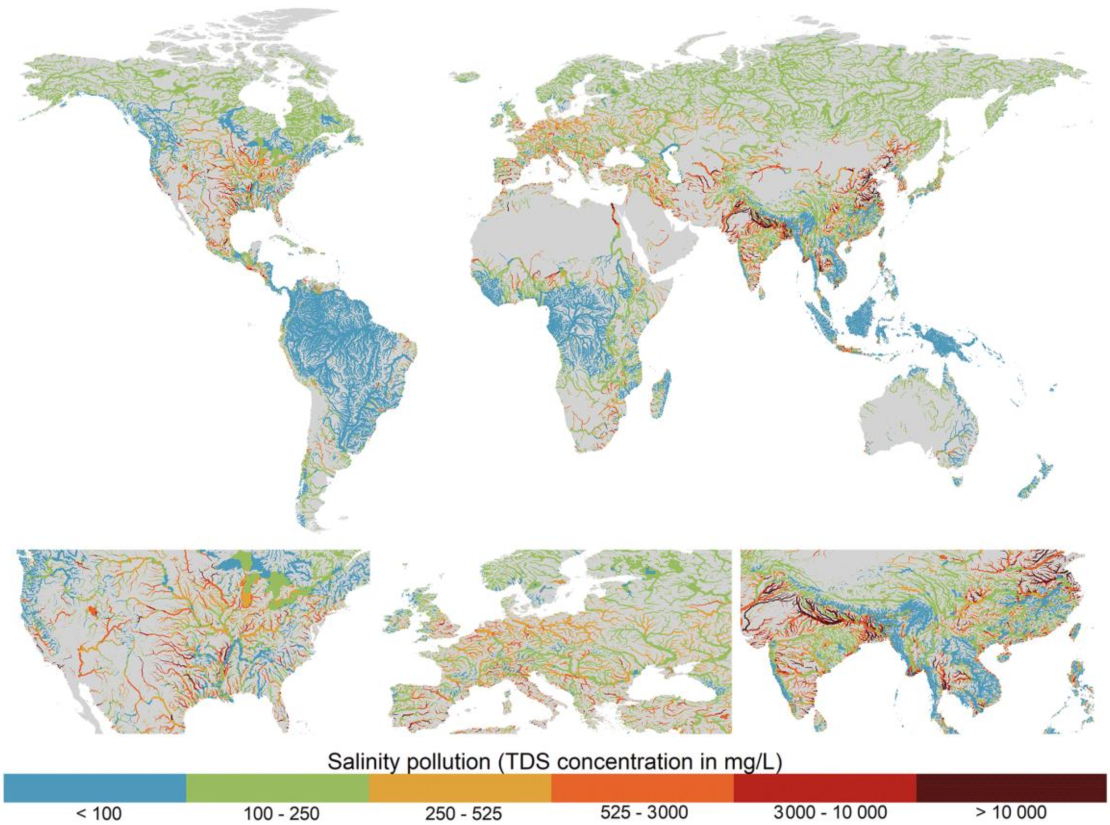
Figure 12: Schematic illustration of PCR-GLOBWB describing interactions, states, and fluxes within and between components (Sutanudjaja et al., 2018).

Table 5: Variables from PCR-GLOBWB (Shen et al., 2022).

Code of variable	Unit	Brief explanation of variable
p	m/day	precipitation
t	°C	temperature
et	m/day	reference potential evapotranspiration
d	–	Julian days (1–365 or 1–366)
baseflow	m/day	baseflow, groundwater discharge
directRunoff	m/day	surface runoff
domesticWaterWithdrawal	m/day	domestic water withdrawal
gwRecharge	m/day	groundwater recharge, fluxes from the lower soil layer to groundwater stores
industryWaterWithdrawal	m/day	industrial water withdrawal
interflowTotal	m/day	interflow, shallow sub-surface flow
irrigationWaterWithdrawal	m/day	water withdrawal allocated for irrigation purposes
livestockWaterWithdrawal	m/day	water withdrawal allocated for livestock demand
nonIrrWaterConsumption	m/day	non-irrigation sectoral (domestic, industry and livestock) water consumption, i.e. non-irrigation sectoral withdrawal minus return flow
snowCoverSWE	m	snow cover/storage in water equivalent thickness (excluding liquid part)
snowFreeWater	m	liquid water/meltwater storage in the snowpack
storGroundwater	m	groundwater storage (renewable part)
storUppTotal	m	S_1 actual upper soil water storage
storLowTotal	m	S_2 actual lower soil water storage
surfaceWaterStorage	m	surface water storage (lakes, reservoirs, rivers, and inundated water)
totLandSurfaceActuaET	m/day	total evaporation and transpiration from land part
totalEvaporation	m/day	total evaporation and transpiration from land and water body parts

DynQual model

The high-spatio-temporal resolution surface water quality model DynQual (Jones et al, 2023) has a spatial resolution of 5 × 5 arcmin (approx. 10 km at the Equator) and provides simulations at a daily time step. The model builds on the framework of DynWat, a global water temperature model that simulates daily water temperature and ice thickness. The DynQual model simulates water temperature (Tw), biological oxygen demand (BOD) to represent organic pollution, concentrations of total dissolved solids (TDS) (Fig.13) to represent salinity pollution and faecal coliform (FC) as an indication of pathogen pollution. This model considers a wide range of socio-economic and hydro-climate drivers. There are two options for running DynQual. The first option is a stand-alone configuration with a specific discharge fed from any land surface or hydrological model. The second option is coupled with the PCR-GLOBWB2 model. Offline runs require both pollutant loading and hydrological input data. While runs coupled with PCR-GLOBWB2 require climate-forcing and socio-economic forcing. It is possible to add loads based on socio-economic, and simulated hydrological data or provide pollutant loadings directly as input data (Jones et al., 2023).



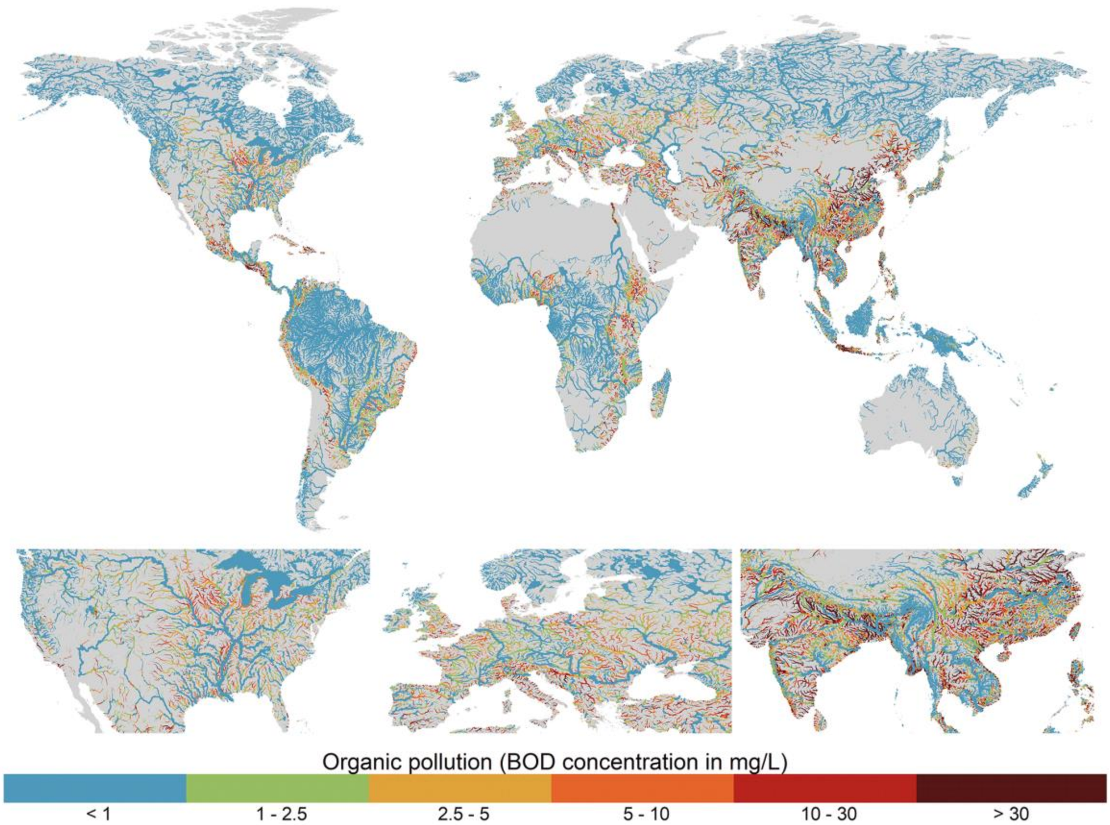


Figure 13: Annual average total dissolved solids (TDS) concentrations and average biological oxygen demand (BOD) concentrations for the period 2010–2019 plotted for rivers with > 10 m3 s−1 annual average discharge. (Jones et al., 2023).

4.2 Analysis

There will be looked to various relations between the physiological characteristics of wetlands to plant species richness. This is achieved by using various statistical methods. First, the minimum, maximum, standard deviation and coefficient of variance are calculated for each dataset, then the following statistical methods are used.

Matrix correlation:
A matrix correlation plot is used to show the correlation coefficients between several variables. The values range from -1 to 1. Here -1 indicates a negative linear correlation between two variables. 0 indicates no linear correlation between two variables and 1 indicates a positive linear correlation (Zach, 2023b).

Regression Analyses:
For this thesis linear, power and inverse regression are used to find the best-fitting relation. Linear regression is used for modelling the relationship between a dependent variable and an independent variable. The aim is to find the best-fitting linear equation that predicts the dependent variable based on the values of the independent variables (Bevans, 2023b). The equation of the linear relationship can be written as:

$$y = b_0 + b_1 \cdot x + \epsilon$$

- y = dependent variable
- b_0= y-intercept
- b_1= slope
- x = independent variable
- ϵ = error term

Inverse regression is used when the relation between two variables of which one variable increases corresponds to a decrease in the other variable. In other words, it is the inverse of a given relation obtained by interchanging the elements of each pair of the given relation (Jafar & Jafar, 2023). If you have a function *f*(*x*), its inverse function is denoted as $f^{-1}(x)$. First are the variables x and y interchanged and then the equation is solved for y. The equation of the inverse relation can be written as (ChiliMath, 2023):

$$y = b_0 + b_1 \cdot \frac{1}{x}$$

- y = dependent variable
- b_0= y-intercept
- b_1= slope
- x = independent variable

Power regression is used when the relation between the variables can be described by an equation where one of the variables is raised to a power involving the other variables. When plotted on a log-log scale, the power relation appears as a straight line. The equation of the power relation can be written as (Zach, 2021a):

$$y = a \cdot x^b$$

- y = dependent variable
- a = constant factor
- x = independent variable
- b = exponent, which determines the power-law relation

In the Scatter plots the number of species is on the y-axis and the physiological characteristics of the wetlands on the x-axis. The axis for the power regression is on a log scale. The p-value, R^2 and the linear regression equations are included in the plots. When the p-value is smaller than 0.05 the relation is statistically significant, this suggests the effect under study likely represents a real relationship rather than just random chance (Zach, 2021b). The R^2 represents how well the statistical model predicts the outcome. The value of R^2 ranges from 0 to 1, with 0 indicating that the model does not explain any of the variability in the response variable, and 1 indicating that the model explains all of the variability (Turney, 2023).

Multiple linear regression (MLR):
Machine learning focuses on the development of statistical models and algorithms. The goal is to develop systems that can learn from data and make decisions or predictions based on that learning. The machine learning is used for the multiple linear regression model (Nikolopoulou, 2023). Multiple linear regression is an extension of linear regression. MLR involves predicting a dependent variable based on the linear relationship with two or more independent variables. This equation of the multiple linear relationship can be written as (Bevans, 2023a):

$$y = b_0 + b_1 \cdot x_1 + b_2 \cdot x_2 + \cdots + b_n \cdot x_n + \epsilon$$

- y = dependent variable
- b_0= y-intercept
- $b_1, b_2, \ldots., b_n$= coefficients for the independent variables
- $x_1, x_2, \ldots., x_n$= are the independent variables
- ϵ = error term

This statistical method is used to determine the best fit between species richness and physiological characteristics. After the MLR the error between the predicted and measured values is calculated. The error percentage is calculated as error divided by measurement times 100%. When the error is smaller than plus-minus 20% the model predicts the values well. To evaluate the performance of the model the K-fold cross-validation is used. The data set is subdivided into training and testing of 5 subsets or folds. The model is trained and evaluated 5 times, using a different fold as the test set in each iteration while the remaining folds are used for training. For each iteration, 4 folds are used for training and the remaining fold is used to test the model (Fig 14) (Zach, 2022). The R^2 and the adjusted R^2 are used as evaluation metrics. For each iteration, the performance values are averaged to obtain a more robust estimate. After the validation, the R^2 can be compared to conclude if the model is overfitting. If the test R^2 is close to the training R^2 the model generalizes well. If the test R^2 is significantly lower, a large gap, than the training R^2, it indicates overfitting. The model is too specific to the training data and does not generalize the new data well. Overfitting is a common problem in machine learning. The model leans the training data too well and captures fluctuations, random patterns and noise, which does not represent the true relationship of the data. it performs well on the training data but fails to new data.

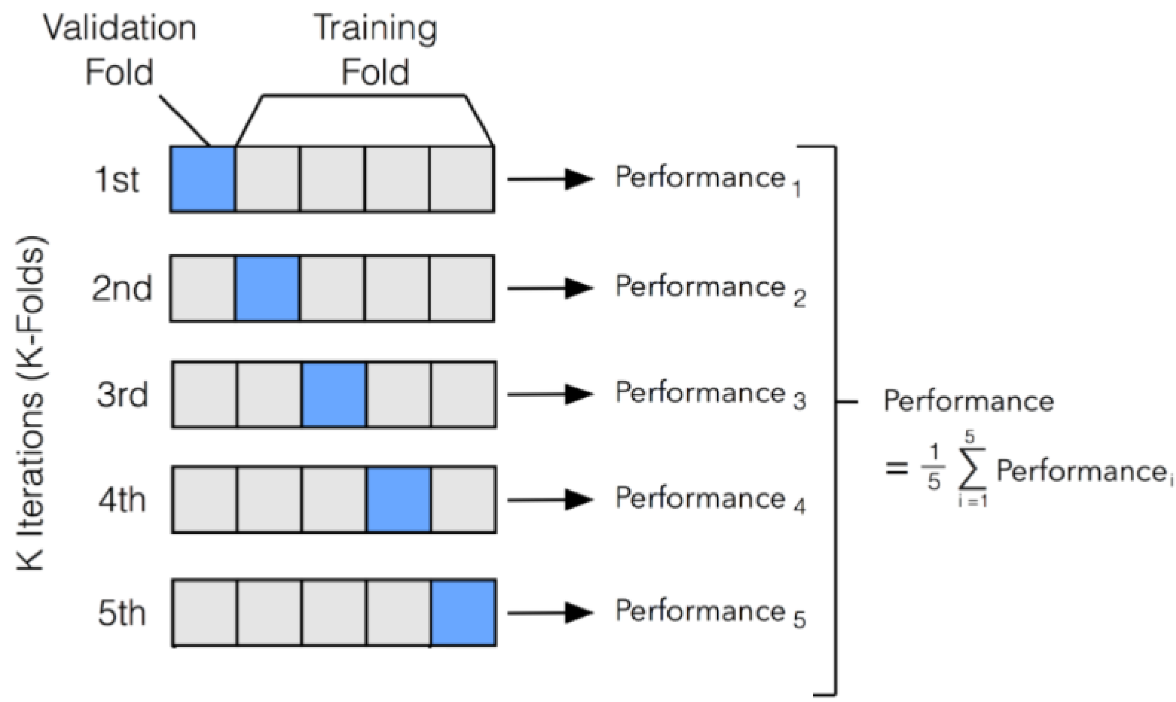


Figure 14: K-fold cross-validation (Sreekrishnan, 2023).

Chapter 5: Results

5.1 General statistics of data

As previously mentioned, England and Australia exhibit distinct differences in climate, which is also reflected in their wetland characteristics. In comparison, England's wetland area is smaller than that of Australia (Tables 6,7). Additionally, Australian wetlands have greater groundwater depth, discharge, and evaporation rates. Moreover, there is a difference in groundwater recharge between the two regions, with Australia recording negative values, whereas England has positive groundwater recharge values. In terms of water quality metrics such as salinity (TDS), Biological Oxygen Demand (BOD), Total Phosphorus (TP), and Nitrates/Nitrites (NOxN), Australian wetlands show lower values than those found in England. The BOD and electrical conductivity (EC) are taken from the World Bank's 'Quality Unknown' dataset, which shows high electrical conductivity values. The BOD from this ML-derived dataset (Damania, 2019; Desbureaux et al, 2022) is much lower in comparison to the BOD from the process-based DynQual model (Jones et al, 2023).

Table 6: Minimum, maximum, mean, standard deviation and coefficient of variation of the variables for England.

Variables	Minimum	Maximum	Mean	Standard deviation	Coefficient of variation
Area (km^2)	0.28	622.12	55.61	108.12	1.94
Groundwater recharge (m/day)	0.00016	0.0032	0.0013	0.00066	0.52
Groundwater depth (m)	0	64.36	10.97	11.33	1.03
Evaporation (m/year)	0.18	0.66	0.34	0.13	0.38
Discharge (m^3/s)	0.044	168.41	7.22	23.63	3.27
Salinity TDS (mg/L) - DynQual	66.40	7761.83	1112.85	1613.72	1.45
BOD (mg/L) - DynQual	1.14	411.87	48.38	92.58	1.91
TP (mg/L) – World Bank Quality Unknown	0.064	0.80	0.39	0.15	0.39
NOxN (mg/L) World Bank Quality Unknown	0.47	5.72	2.75	1.12	0.41
BOD (mg/L) World Bank Quality Unknown	1.48	2.95	2.11	0.27	0.13
Salinity EC(mS/cm^2) World Bank Quality Unknown	86.25	601.58	363.69	135.92	0.37

Table 7: Minimum, maximum, mean, standard deviation and coefficient of variation of the variables for Australia.

Variables	Minimum	Maximum	Mean	Standard deviation	Coefficient of variation
Area (km2)	0.005	19800	350.15	14714	4.20
Groundwater recharge (m/day)	-1.30E-05	0.0084	0.00049	0.0010	2.06
Groundwater depth (m)	0	498.34	35.16	59.70	1.70
Evaporation (m/year)	0.15	1.67	0.59	0.30	0.50
Discharge (m3/s)	0	798.29	28.78	115.27	4.01
Salinity TDS (mg/L) - DynQual	1.02	3370.13	200.17	413.60	2.07
BOD (mg/L) - DynQual	0	112.017	2.45	10.98	4.47
TP (mg/L) – World Bank Quality Unknown	0.027	0.36	0.16	0.068	0.42
NOxN (mg/L) World Bank Quality Unknown	0.11	1.21	0.44	0.17	0.39
BOD (mg/L) World Bank Quality Unknown	0.40	2.48	1.33	10.98	8.27
Salinity EC(mS/cm2) World Bank Quality Unknown	66.11	981.66	407.61	239.56	0.59

The correlation matrix (Fig. 15) shows that species richness and area have the highest correlation. For England, the correlation is 0.41 and for Australia is 0.36. In England, the evaporation (0.13) and the BOD (0.18) show a positive correlation with species richness. While the evaporation (0.2) is only positively correlated for Australia. The correlation between the BOD derived from DynQual and the BOD derived from Quality Unknown is 0.13 (England) and 0.032 (Australia) which is low for the same variable from different datasets. The same can be concluded for the salinity and the electrical conductivity. The correlation is 0.11 (England) and 0.086 (Australia).

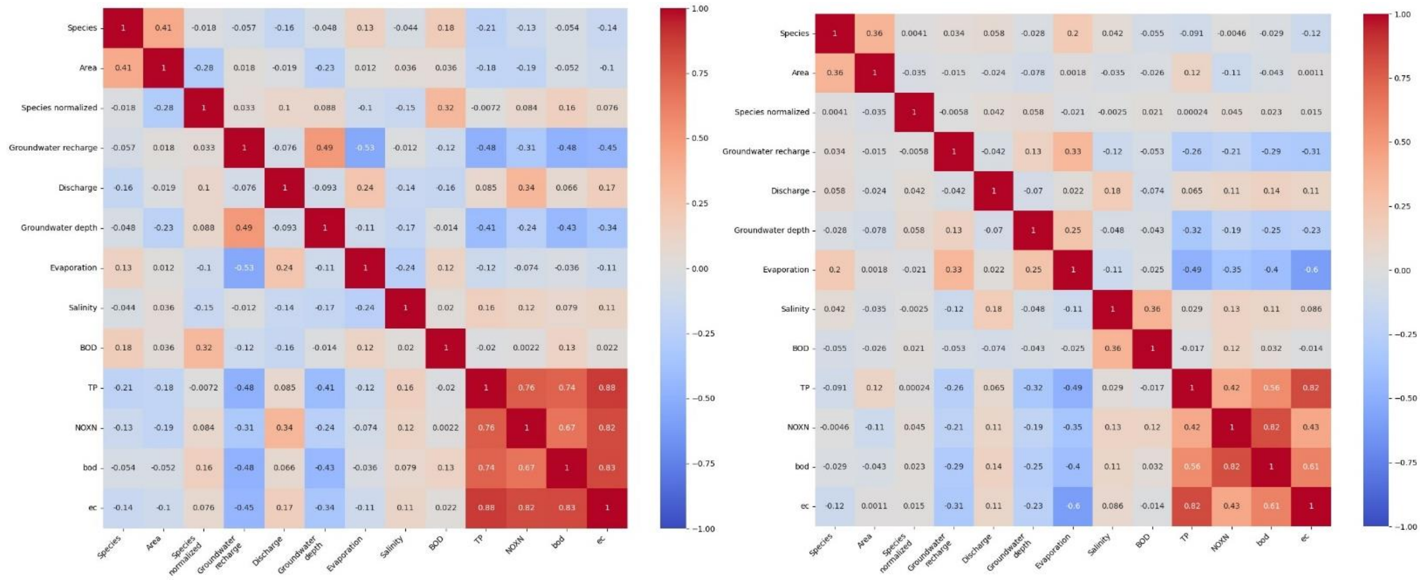


Figure 15: Correlation matrix a) England b) Australia.

5.2 Regression analyses

The regression analysis shows that the area is the best-fitted relationship. Particularly, the power regression shows the highest R^2 and the lowest P-value (Table 8,9) and a high correlation of 0.50 (Fig. 16a) for England and 0.23 for Australia (Appendix 5,6). Furthermore, the power regression proves to be a good fit for the BOD in England, with a slightly lower R^2 and higher P-value (Table 8,9) and a correlation of 0.42 (Fig. 16b). An increase in both area and BOD correlates with an increase in species richness. However, the water quality data for Australia does not well predict the different regression analyses. The same can be concluded over salinity and NOxN for England. The TP in Australia and England show a higher R^2 for the linear regression and is significant for Australia (Fig. 17a). An increase in TP correlates with a decrease in species richness.

The groundwater recharge in both countries and groundwater depth in Australia show a slightly better fit for the inverse regression (Fig. 17b). There is a slight increase in R^2 and the regression coefficient is significant. The species decreases when the groundwater depth and recharge increase. The p-value for evaporation and discharge is low, indicating significant relationships for Australia while this is not observed for England, possibly due to lower values (Table 6,7). The discharge has a higher R^2 in the power regression, whereas the evaporation has a higher R^2 for the linear regression. Consequently, an increase in both discharge and evaporation corresponds to an increase in species richness. The regression analyses are provided in Appendix 1 to 6.

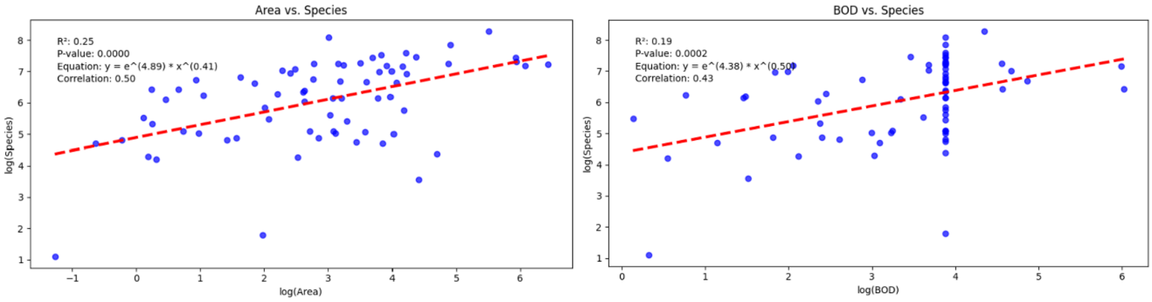


Figure 16: Power regression England a) area vs species b) BOD vs species.

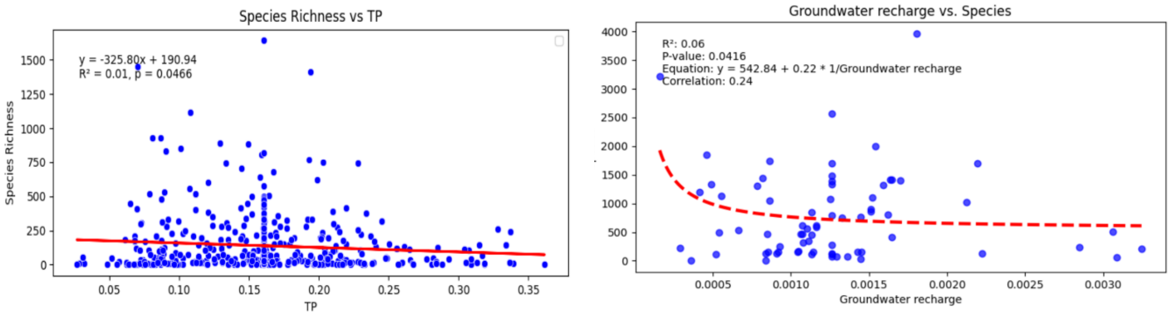


Figure 17: a) linear regression Australia TP vs species b) Inverse regression England Groundwater recharge vs species.

Table 8: The R^2 and P-value for the linear, inverse and power relation England.

Variables	R^2 linear	R^2 inverse	R^2 power	P-value linear	P-value inverse	P-value power	Regression coefficient linear	Regression coefficient inverse	Regression coefficient power
Groundwater recharge	0.00	0.06	0.00	0.6361	0.0416	0.8205	-71666.08	0.22	0.07
Area	0.17	0.09	0.25	0.0004	0.0113	0.0000	0.05	-440.64	0.41
Groundwater depth	0.00	0.00	0.01	0.6883	0.7247	0.5386	-3.28	0.00	-0.02
Evaporation	0.02	0.03	0.00	0.2684	0.1511	0.5615	842.84	-128.76	0.28
Discharge	0.03	0.01	0.00	0.1705	0.3381	0.8406	-5.67	-23.01	0.02
Salinity	0.00	0.04	0.02	0.7168	0.0957	0.1934	-0.03	-56420.45	0.23
BOD	0.03	0.07	0.19	0.1414	0.0232	0.0002	2.03	-1302.09	0.50
TP	0.04	0.00	0.02	0.0805	0.7858	0.1885	-1168.21	10.35	-0.45
NOxN	0.02	0.00	0.02	0.2717	0.8620	0.2663	-100.87	-48.15	-0.39

Table 9: The R^2 and P-value for the linear, inverse and power relation Australia.

Variables	R^2 linear	R^2 inverse	R^2 power	P-value linear	P-value inverse	P-value power	Regression coefficient linear	Regression coefficient inverse	Regression coefficient power
Groundwater recharge	0.00	0.01	0.00	0.4527	0.0327	0.2281	7337	0.00	0.00
Area	0.13	0.01	0.23	0.0000	0.0372	0.0000	0.05	-1.33	0.79
Groundwater depth	0.00	0.04	0.02	0.5406	0.0000	0.0063	-0.10	0.00	-0.10
Evaporation	0.04	0.03	0.02	0.0000	0.0002	0.0007	148.35	-31.76	0.05
Discharge	0.00	0.01	0.03	0.2072	0.1162	0.0002	0.11	0.00	0.42
Salinity	0.00	0.00	0.02	0.3579	0.6773	0.1021	0.04	-43.13	0.12
BOD	0.00	0.00	0.01	0.2266	0.8359	0.1697	-1.78	0.00	0.19
TP	0.01	0.00	0.00	0.0466	0.7729	0.8509	-325.80	0.72	0.00
NOxN	0.00	0.00	0.01	0.9204	0.9194	0.1434	-6.61	-1.00	0.02

5.3 Multiple Linear Regression

For the multiple linear regression (MLR) 32 out of the 71 wetlands are used for England and 125 out of 479 wetlands are used for Australia. Fewer wetlands were used because this technique excludes the wetlands with a missing variable. The predictions of the MLR model for England are closer to the measurement values compared to those for Australia (Fig. 18). The R^2 for England is notably higher at 0.76 compared to 0.30 for Australia. Some p-values become more significant using the MLR compared to the linear regression used in section 5.2. For England the NOxN, TP, discharge, groundwater recharge and area get significant and the p-value for salinity and groundwater depth get smaller/improved. For Australia the NOxN and salinity get significant and the BOD and groundwater recharge get smaller/improved.

When performing MLR solely for the area, the R^2 for England is 0.66 and 0.26 for Australia. This indicates that using one predictor variable is insufficient to explain species richness. Using the MLR with a log-log scale will decrease the R^2 for England to 0.46 and the R^2 for Australia will increase to 0.56 (Fig. 19). The discharge and BOD for Australia get more significant using the log-log scale (Table 10,11) which corresponds to the power regression analysis (Table 9). Similar improvements are observed for the BOD in England. The area is significant for both the anal.

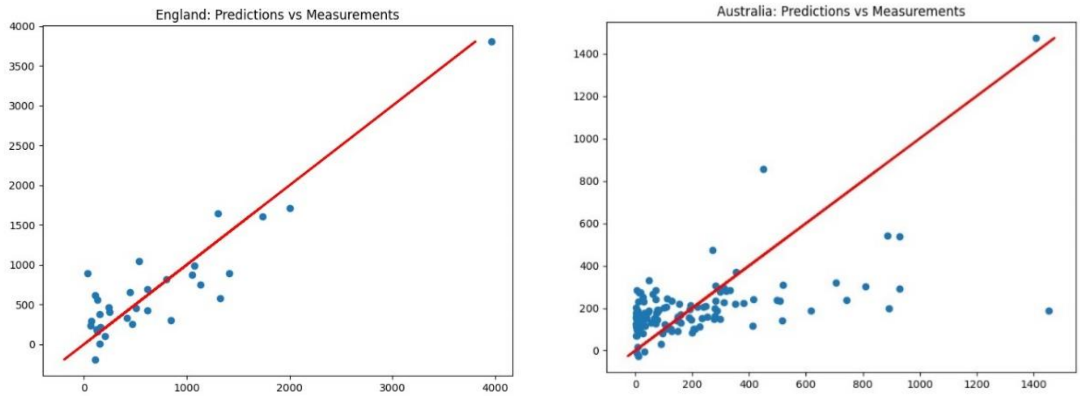


Figure 18: Predictions vs measurements from the multiple linear regression.

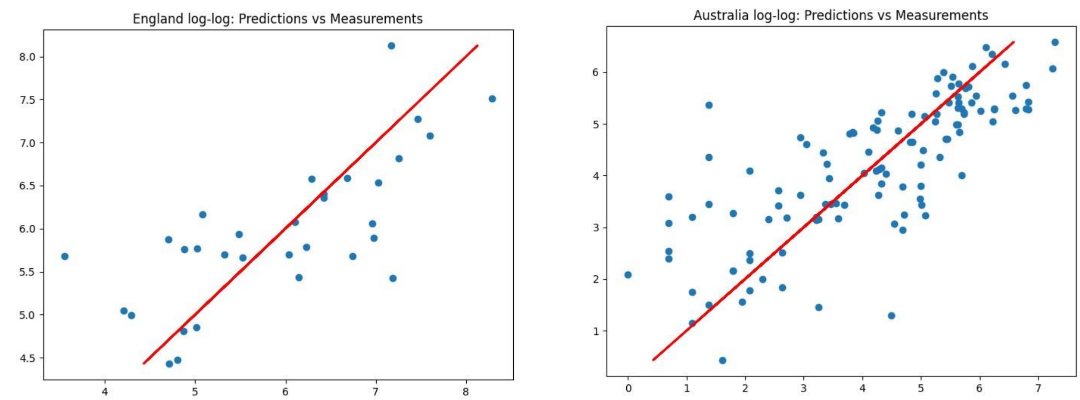


Figure 19:Predictions vs measurements from the multiple linear regression on log-log scale.

Table 10: Regression coefficient and P-value for England and Australia using the MLR.

Variables	Regression coefficient England	P-value England	Regression coefficient Australia	P-value Australia	Regression coefficient Australia (small area)	P-value Australia (small area)
Groundwater recharge	-4.38+05	0.046	-2.86E+04	0.322	-1.92E+04	0.311
Area	13.41	0.000	0.097	0.000	4.41	0.000
Groundwater depth	7.14	0.431	-0.18	0.612	0.082	0.713
Evaporation	-233.48	0.801	146.09	0.071	64.31	0.277
Discharge	-9.59	0.008	-0.092	0.492	-0.0073	0.925
Salinity	-0.094	0.053	0.11	0.035	0.088	0.003
BOD	-0.13	0.877	-3.92	0.069	-1.98	0.109
TP	-2869.28	0.039	-697.52	0.223	963.29	0.017
NOxN	310.85	0.022	289.00	0.033	297.56	0.001

Table 11: Regression coefficient and P-value for England and Australia using the MLR on a log-log scale.

Variables	Regression coefficient England	P-value England	Regression coefficient Australia	P-value Australia	Regression coefficient Australia (small area)	P-value Australia (small area)
Groundwater recharge	-0.73	0.312	0.020	0.392	0.020	0.538
Area	0.33	0.006	0.44	0.000	0.29	0.010
Groundwater depth	-0.089	0.77	0.067	0.530	-0.18	0.389
Evaporation	-0.46	0.54	0.21	0.538	0.081	0.880
Discharge	-0.15	0.45	0.14	0.009	0.11	0.200
Salinity	-0.34	0.14	0.069	0.349	0.090	0.372
BOD	0.37	0.088	0.14	0.007	0.070	0.372
TP	-1.39	0.17	-0.42	0.257	-0.048	0.389
NOxN	0.80	0.36	0.73	0.105	0.78	0.220

When the hydrological and water quality values are separated in the multiple linear regression (MLR), there is a decrease in the R^2. If the Area is excluded the R^2 becomes negative, therefore the Area is the most important variable in the MLR. In England, excluding BOD, evaporation, and groundwater depth leads to an increase in R^2 to 0.78. For Australia, the R^2 increases to 0.31 when the groundwater depth and discharge are excluded. In contrast to England, the BOD and evaporation are important values in the MLR for Australia. The same is found for the discharge in England.

The analysis was revised for wetlands with an area smaller than 100 km^2, aligning with the size of one grid cell in the DynQual and PCR-GLOBWB model. This revision applied exclusively to Australia, as wetlands in England were already smaller than 100 km^2. 46 wetlands were excluded and 79 were used in the analysis. The R^2 increased from 0.30 to 0.46, accompanied by a decrease in both the root mean square error and mean absolute error (Fig. 20). The R^2 increases to 0.48 when the groundwater depth, groundwater recharge and discharge are excluded. When the analysis with small is used in MLR on a log-log scale the R^2 decreases to 0.28 and the p-value does not improve.

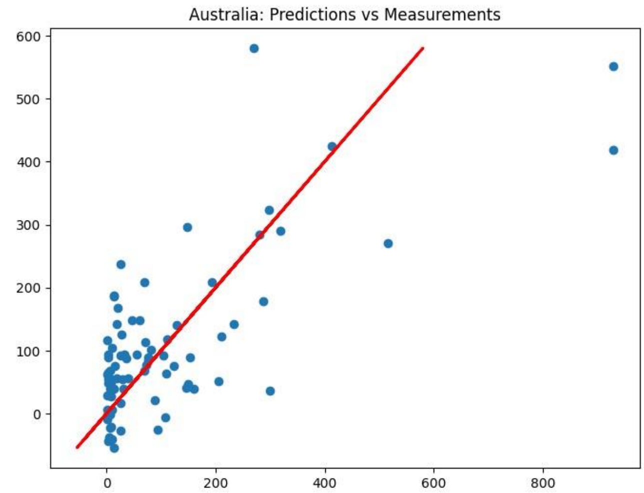


Figure 20: Predictions vs measurements from the multiple linear regression for only small areas.

About 34% of England's predicted values deviate less than 20% from the actual measurements, whereas Australia shows a percentage of 14.4. When the analysis is revised for the smaller areas the percentage of Australia increases to 16.5%.

The cross-validation shows that for England one of the five folds is overfitted. The average relative difference is 22.2%, which is quite low. Hence, the MLR model for England is deemed to accurately represent the true relationship of the data and is expected to perform well on new data. The cross-validation of Australia indicates overfitting, with an average relative difference of 84.0%, signifying a high degree of variability. This suggests that the MLR model for Australia captures fluctuations, random patterns, and noise, rather than representing the true relationship. By considering fewer variables, such as excluding groundwater depth, recharge, and discharge, the fit for Australia improves slightly, with the average relative difference decreasing to 80.2%. However, for England, excluding BOD, evaporation, and groundwater depth does not enhance the fit. Furthermore, reducing the wetland area to 100 km2 results in a decrease in the average relative difference to 51.9% for Australia, though it remains relatively high. The averaged R^2 for the tested data is 0.67 for England and 0.26 for Australia (Table 12). This is lower compared to the original R^2 in the MLR (England 0.755 and Australia 0.30). The tested averaged R^2 is also lower compared to the average R^2 of the trained data.

Table 12: cross-validation results, R^2 train and R^2 test and the relative difference between them for England and Australia.

i	R^2 train England	R^2 test England	relative difference (%)	R^2 train Australia	R^2 test Australia	relative difference (%)
1	0.794344023	0.835953023	5.238158628	0.499530076	0.001310198	99.73771379
2	0.866078368	0.662714772	23.48096924	0.231927969	0.683607993	194.7501311
3	0.85971609	0.324338109	62.27381197	0.355886427	0.156579942	56.00283417
4	0.871130674	0.723676639	16.92674118	0.366612864	0.262567598	28.38014601
5	0.822574994	0.795702547	3.266868933	0.353826388	0.206817847	41.54821302
Average	0.84276883	0.668477018	22.23730999	0.361556745	0.262176715	84.08380761

Chapter 6: Discussion

6.1 Analysis

The species-area relation shows the best-fitted relation of this analysis. The power function for the area has the highest fit which is consistent with the case study of Peintinger et al. (2003) in chapter 2.7. This is also supported by Dengler (2009) who used theoretical considerations and empirical results, to suggest that the power law should be used to describe and compare any type of species-area relation. The power relation is not always the superior fit because some of the variables have a better fit for the inverse or linear regression. In the multiple linear regression (MLR) models the area is the most important variable, which is probably due to the high correlation with species richness.

The MLR model shows that using only one predictor variable (area) is insufficient to explain species richness and therefore including hydrological and water quality variables improves the model fit to species richness. The R^2 improves from 0.17 by only using the area in linear regression to 0.76 including all variables (MLR) for England and from 0.13 to 0.30 for Australia. Using more variables increases the fit because it provides more information about the range of variability and can reduce errors. In addition, some variables may influence both the predictor and response variables but are not of primary interest. Including these variables in the analysis helps control for their effects, ensuring that the relationship between the primary predictor variables and the response variable is accurately estimated. Using more variables shows that the P-values are lower (more significant) compared to using one variable. When only the area was used in the MLR the R^2 (0.66 England) is higher compared to the R^2 (0.17 England) of the linear regression. This should be the same but the MLR uses fewer wetlands and therefore the accuracy is less.

Using the MLR with a log-log scale will decrease the R^2 for England and increase the R^2 for Australia. The area and organic pollution as indicated by biochemical oxygen demand (BOD) concentrations show a better fit for the power regression. I anticipated that using a log-log scale in the MLR would increase R^2 in the analysis because the area is the most important driver of species richness. For England fewer wetlands (only 32) are used compared to Australia, this can also influence the outcome because less information is available to use in the MLR. In the analyses, the species normalized (species/area) was used but eventually, it was left out due to a lower R^2 compared to the R^2 of species richness.

The groundwater depth, recharge, BOD, evaporation and discharge are not as important in the MLR. Discharge, evaporation and groundwater recharge are important factors influencing groundwater level (Condon et al., 2021). The average groundwater depth for wetlands in England is 10.97 m and in Australia 35.16 m which is deep. The groundwater depth in wetlands can range from a few centimetres to several meters (Wu et al., 2020). The groundwater depth is very deep hence not having an impact on the wetland. Assuming the same for recharge and discharge if the water table is too deep. In addition, the evaporation, groundwater recharge and discharge are not strongly correlated to species richness and therefore will not influence the MLR much. A reason for this is that the variables do not fluctuate much and stay constant over the wetland size and area. The mean and the range of evaporation, groundwater recharge and discharge are relatively small for England and Australia. Excluding the BOD in the MLR for England contradicts what I expected due to the high correlation to species richness and a good fit with the power regression. The BOD values are higher compared to Australia and for Australia, the BOD is important in the MLR. You should expect it the other way around due to Higher pollution in England.

In the regression analysis, the salinity does not show a strong relation. This is the opposite of what I expected. The case study showed that an increase in salinity causes a decrease in plant species richness because most plants have built a tolerance, which depends on the type of plant, and when this is exceeded, the plant can be altered or disappear. High salt concentrations cause nutrient imbalance which affects the nutrient uptake. High concentrations in the plant are toxic and cause alterations in water relations. It will alternate some important processes in the plant and result in restriction of growth, development and productivity. Some plants evolved to enable survival skills under saline conditions (Bernstein, 2019). No distinction is made in what kind of species are found in the wetlands. Maybe some species have developed some of the survival skills and therefore no strong relation is found or the tolerance is not exceeded. The case study of Antonellini & Mollema (2010) Shows a strong decrease in plant species above 3000 mg/L. The mean salinity in the wetlands of Australia and England is much lower (1112 and 200 mg/L). Only a maximum of 4 wetlands is above the threshold.

The electrical conductivity values are too high to be realistic If you convert EC from mS/cm^2 roughly to mg/L it shows much larger values than the salinity of the PCR-GLOBWB model. These values are unrealistic for terrestrial wetlands. Normal EC values are between 10-25 mS/cm and the values of the Quality unknown data are between 86 and 600 mS/cm^2. If you multiply the area to get mS/cm the values will become larger. The high values be explained by the coarse resolution of the Quality unknown data (0.5 deg, ~50x50 km gridcell size). Averaged values for EC are taken over a large area, which does not represent the EC for the small wetland. The averaged values mostly are higher due to coastal salinity and therefore the EC is excluded from the analyses.

The water quality variables such as Biological Oxygen Demand (BOD), Nitrate/Nitrite (NOxN) and Total Phosphorus (TP) are lower in Australia than in England. This is due to lower population density in Australia. The population density for England is 434 people per square kilometre (Park, 2022) and for Australia 3.42 people per square kilometre (Macrotrends LLC, n.d.). A lower population density causes less pollution and therefore the water quality data for Australia is lower than for England. The correlations with the water quality data are much lower for Australia. This is probably due to much lower values and therefore gives weaker relations.

The increase in BOD causes an increase in species richness which contradicts what I expected. An increase in BOD indicates an increase in the amount of organic matter present in water. More dissolved oxygen is needed to degrade dissolved organic content in the water, this leaves plant species with less dissolved oxygen which is essential for plant growth and survival of aquatic plants (Water Science School, 2018b). However, the relationship between BOD and species richness is not straightforward and can be influenced by other factors such as nutrient enrichment, temperature, and hypoxia (Vipin, 2023). The BOD range of the Quality unknown data is much smaller than the BOD of the PCR-GLOBWB. The values of the PCR-GLOBWB are higher which indicates a poorer water quality than the Quality unknown data. 12 wetlands in England are of poor quality, while in the Quality unknown data states that all wetlands are very clean. The Quality unknown data is less reliable for this research because average values are taken for a larger area. The grid cell of Quality unknown is equal to ~50 km resolution while the PCR-GLOBWB model uses grid cells with 10 x 10 km resolution. Therefore the values in the Quality unknown data are less specific for one wetland and include more variation than the PCR-GLOBWB model.

For TP a decreasing correlation is found. Phosphorus tends to attach to soil particles and moves from runoff or groundwater flows into surface water due to urban and agricultural settings. The phosphorus affects the water quality of surface water. TP is an essential element for plant life. Too much of it can accelerate eutrophication, which causes a reduction in dissolved oxygen in water bodies due to an increase of mineral and organic nutrients in rivers and lakes. In the beginning, an increase in TP can boost plant growth. However, when TP increases the BOD increases to degrade the organic matter. Less dissolved oxygen is left for plants which affects plants negatively (Water Science School, 2018c). Nitrate/nitrites are also essential elements for plant life. The overstimulation of aquatic plant and algae growth, resulting from excess nitrogen, can lead to various issues. These include the clogging of water intakes, the depletion of dissolved oxygen during decomposition, and the obstruction of light penetration to deeper waters (Water Science School, 2018a). This causes a negative impact on species richness, however, the regression analyses did not well predict this effect.

6.2 Limitations and implications

Some limitations are encountered during this study. The model is experiencing overfitting, which means that the model is performing well on the training data but shows a lower performance in representing new, unseen data. In such cases, relying on the observation data instead of model output data might be more reliable for evaluating and making decisions.

During data collection, the variables in the PCR-GLOBWB and DynQual model were taken for one coordinate, the middle of the wetland. The variables are not equally distributed over the whole wetland, which was assumed by this approach. For some wetlands multiple grid cells were present but only one grid cell is used in the analysis. In some cases, the values are underestimated by this approach because the middle of the wetland is relatively less under influence by for example pollutants than further to the edge, which can be closer to a pollutant source. This is supported by the approach for smaller areas (one grid cell) in the MLR. If a wetland is smaller or equal to one grid cell it is used in the MLR and the R^2 increases. The relation is better fitted. For the next research, it is therefore advised to take all grid cells within the wetland area and average the values, to get a more representative value for the whole wetland

By using the MLR approach a part of the wetlands are deleted because not a number (Nan) values were present. This is a limitation of the model. Therefore, fewer wetlands were used for the MLR than in the other analysis which can cause a loss of information and affect the accuracy.

Implications for further research can be to use other regression analyses such as exponential and asymptotic which are also mentioned by Dengler (2009). This can be done to see if the variables that show no strong correlation in this analysis have a better fit to other regression analyses. If no better fit is found with the other regression analysis the conclusion of certain variables less influencing the species richness will be supported. Other machine-learning techniques for example nonlinear regression can be used to explore if there is a better fit possible for hydrological and water quality data to species richness. Another suggestion is to couple the spatial distribution of the wetland to various maps for example climate, land use and land use change to determine how the variables are influenced.

Chapter 7: Conclusion

The main aim of this thesis was to investigate the best-fit relationship between the physiological characteristics of inland wetlands and plant species richness in Australia and England. The thesis included a comprehensive literature review about wetlands, plant species richness and drivers impacting wetlands and species richness. In addition, statistical analyses were conducted, using data of hydrological variables, wetland extent and water quality collected from field monitoring and existing models such as PCR-GLOBWB hydrological model and DynQual global surface water quality model. Best-fit relations were explored by various statistical analyses such as a cumulative distribution function, matrix correlation, regression analysis (linear, inverse and power function) and multiple linear regression. Two main research questions were addressed

RQ1) What are the main drivers impacting species richness in wetlands?

The literature review shows that the direct drivers of degradation and loss of rivers, lakes and wetlands and therefore species richness include infrastructure development, land conversion, changes in water use and availability due to for example water withdrawal, pollution, disease control, overharvesting and overexploitation, and the introduction of invasive alien species. The indirect drivers of wetland loss or degradation have been increasing economic development and population growth. Global climate change will exacerbate the pressure on wetlands, which will increase the loss and degradation of many wetlands. It has increased the potential for irreversible change in the ecological state such as the loss or decline of species richness. The statistical analysis shows that the most important driver of species richness is the wetland area.

RQ2) What is the relation between wetland site characteristics in particular discharge, wetland area and water quality, such as organic pollution indicated as biological oxygen demand (BOD), nutrients such as total phosphorus (TP) and nitrates/nitrites (NOxN) and salinity (indicated as TDS) in sustaining wetland plant species richness?

The wetland area is the primary driver of species richness. The power function shows the best fit for species versus area. However, using one predictor variable is insufficient to explain species richness. Therefore hydrological and water quality variables must be included. Decreasing the wetland area in the analysis increases the fit due to some limitations in the approach. The groundwater depth, recharge, organic pollution (BOD), evaporation and discharge are not as important in sustaining wetland plant species richness as the area.

For further research, it is effective to also examine how the wetland variables will change under pressures such as climate change and land-use change and how this will affect the plant species richness. It is then possible to assess the human impacts on wetlands and plant species richness. This makes decision-making easier on how to improve biodiversity in strategic environmental assessments.

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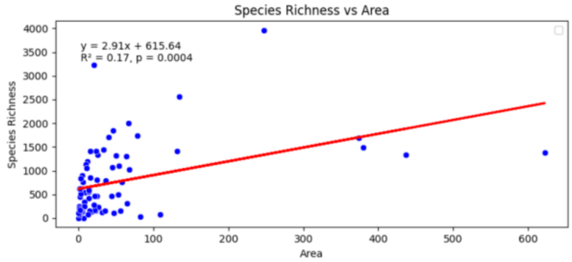
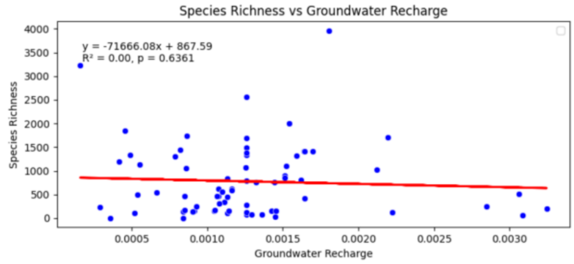
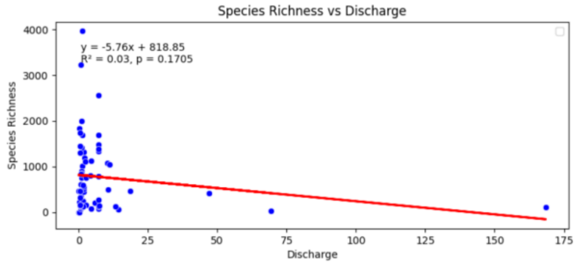
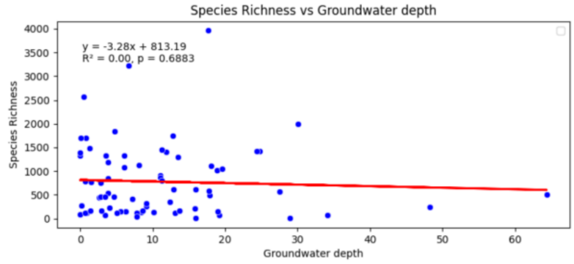
Appendix:

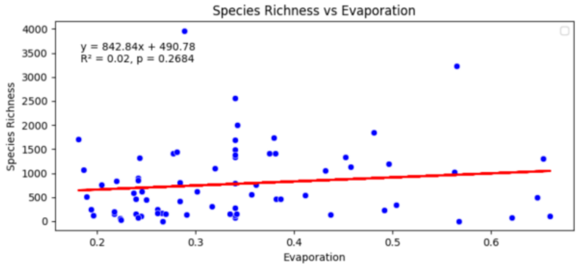
Appendix 1: Linear regression England

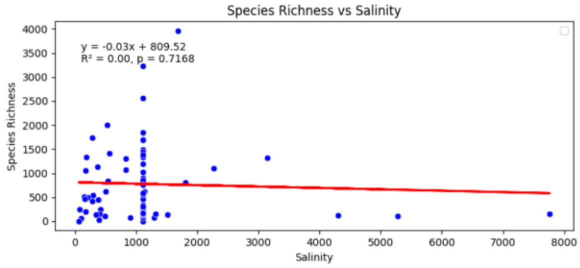


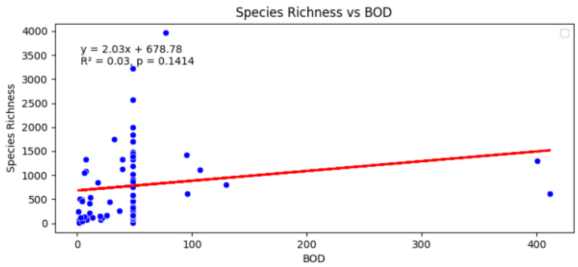


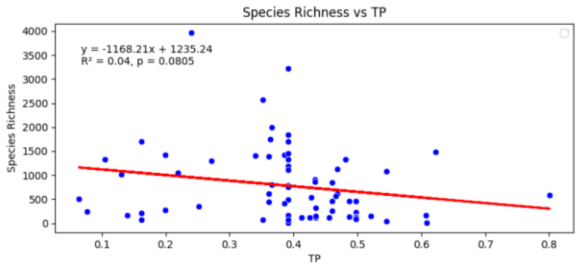


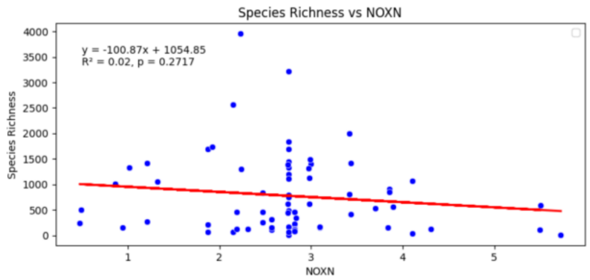




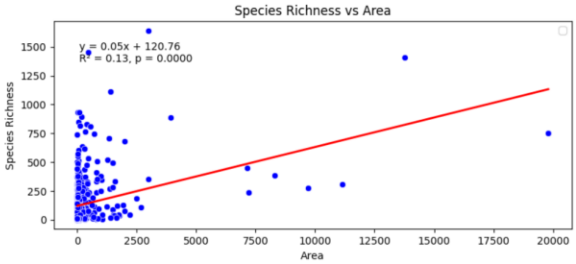


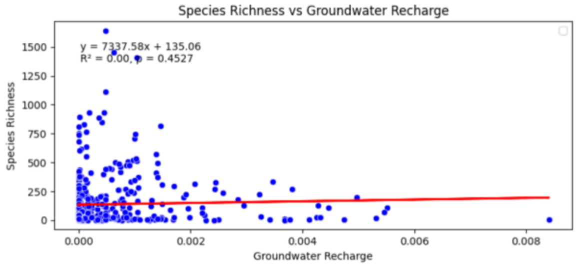


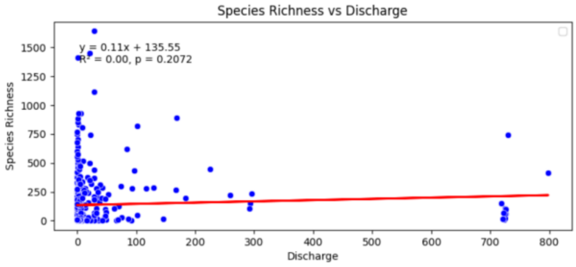


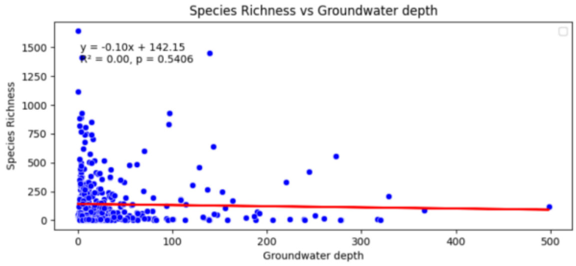


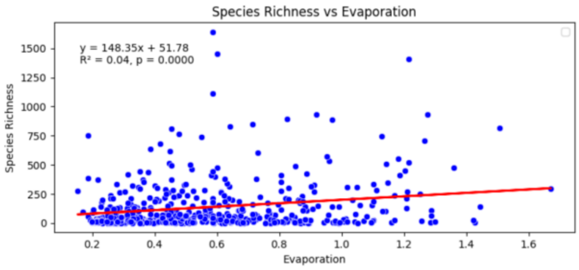
Appendix 2: Linear regression Australia

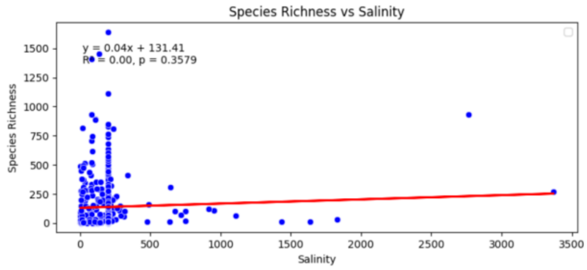


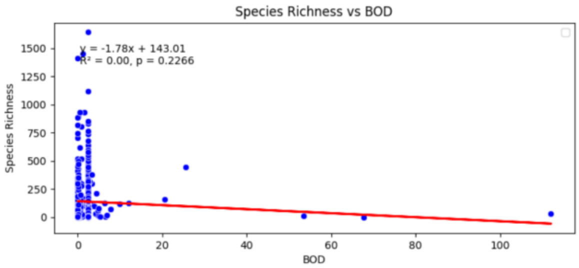


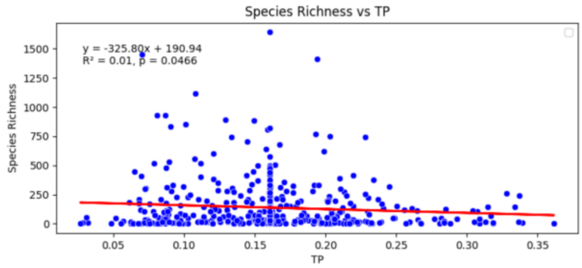


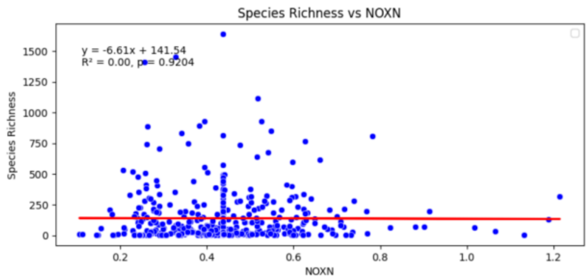




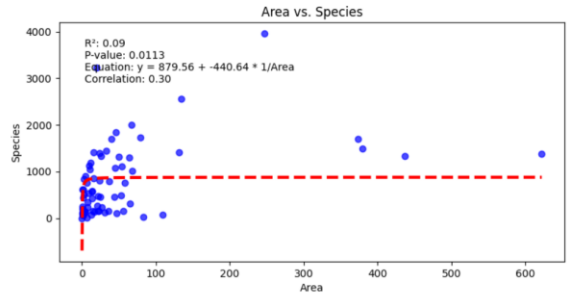


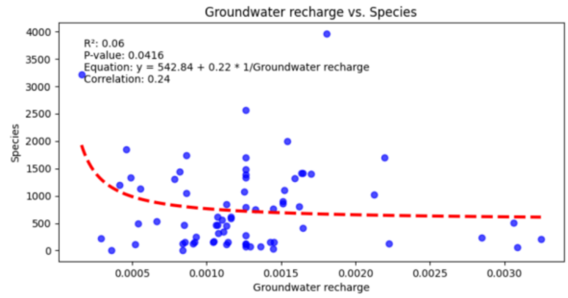


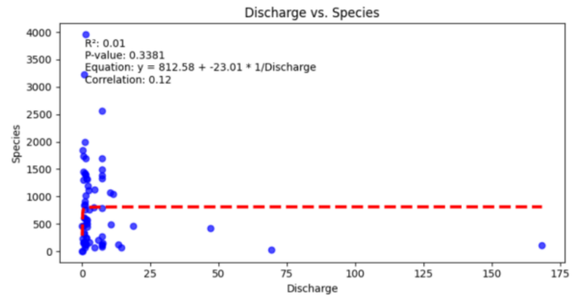


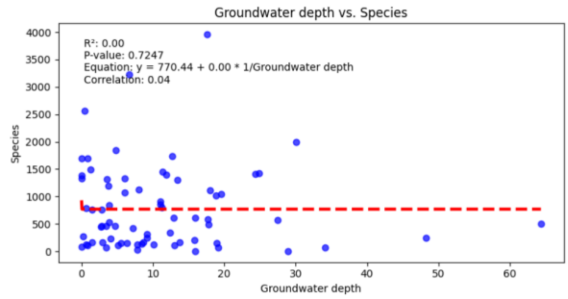


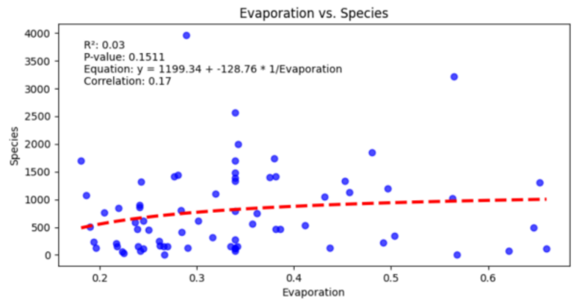
Appendix 3: Inverse Regression England

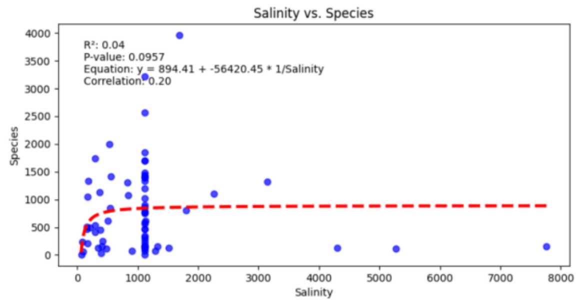


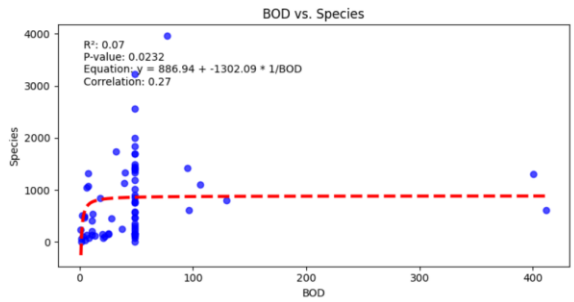


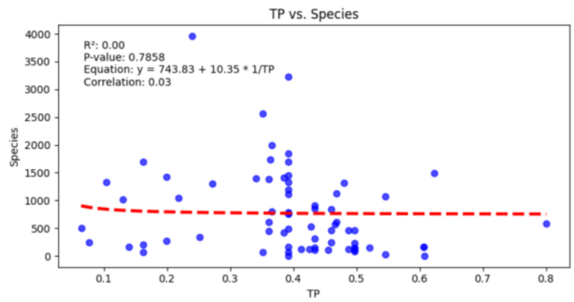


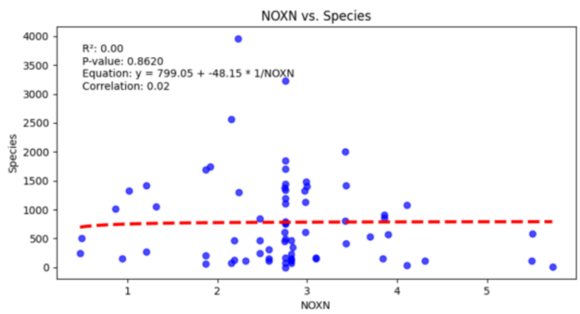




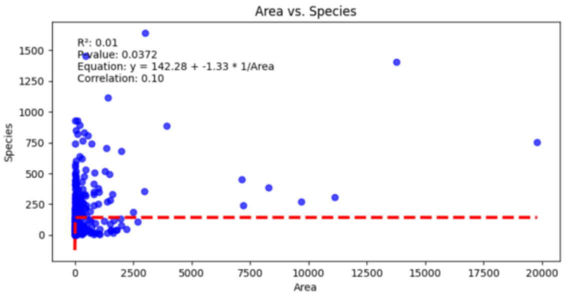


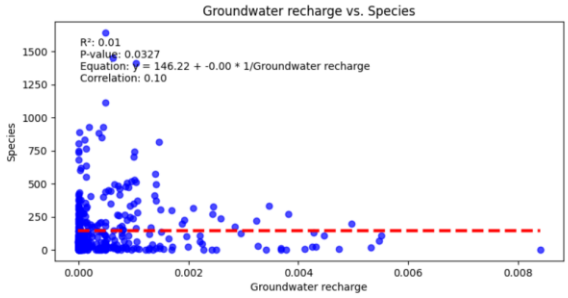


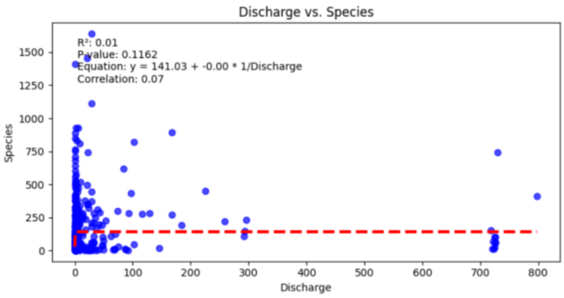


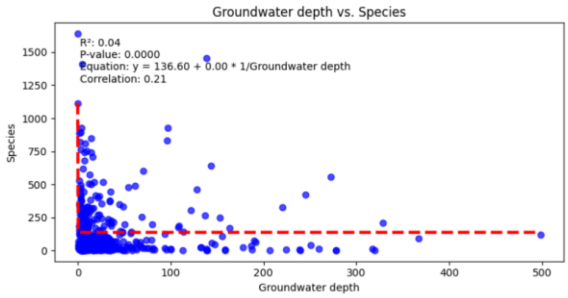


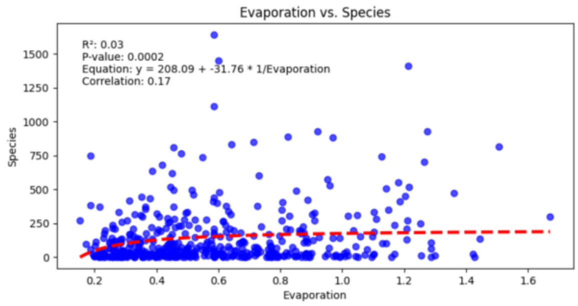
Appendix 4: Inverse Regression Australia

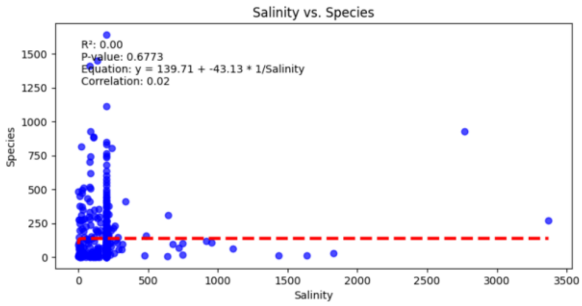


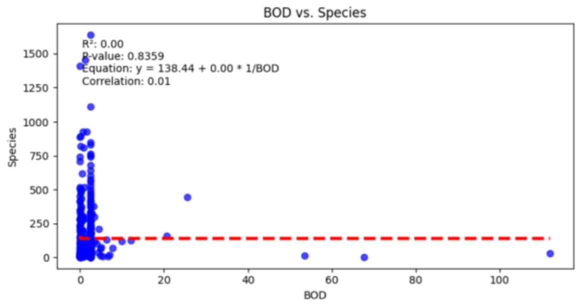


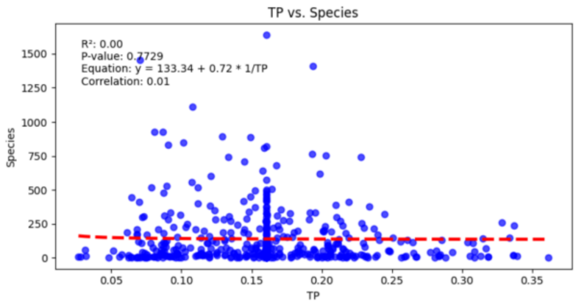


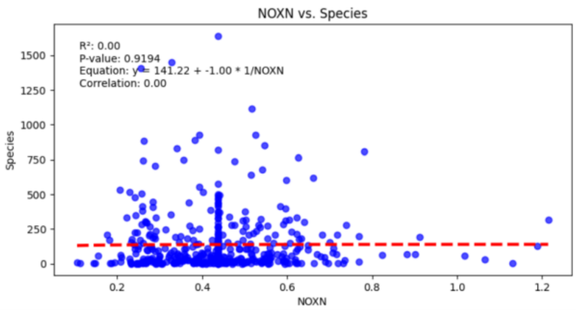




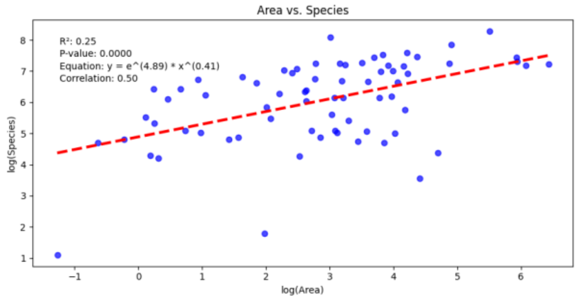


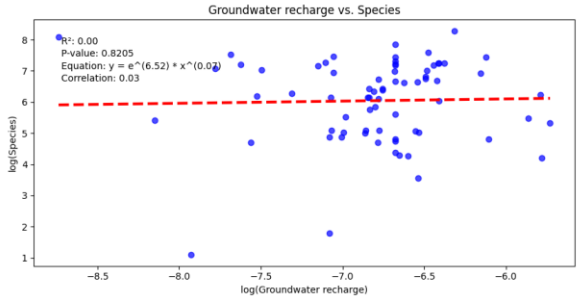


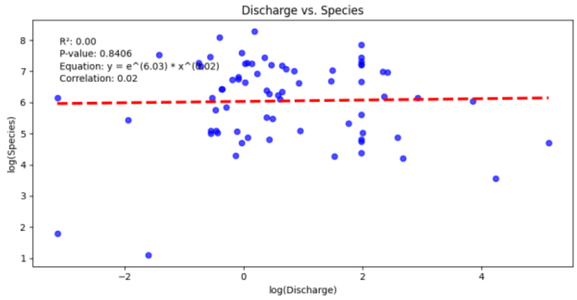


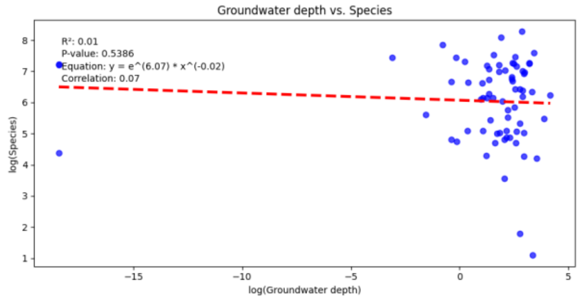


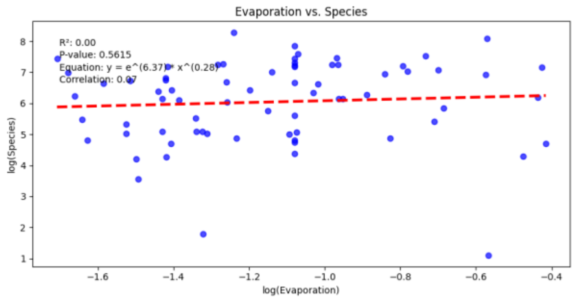
Appendix 5: Power Regression England

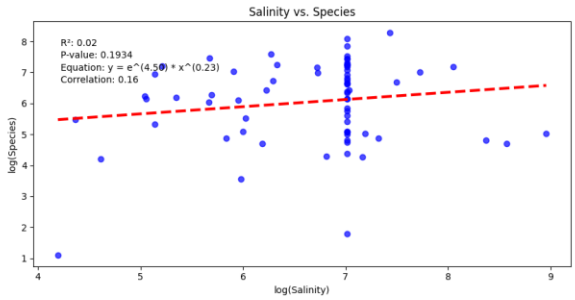


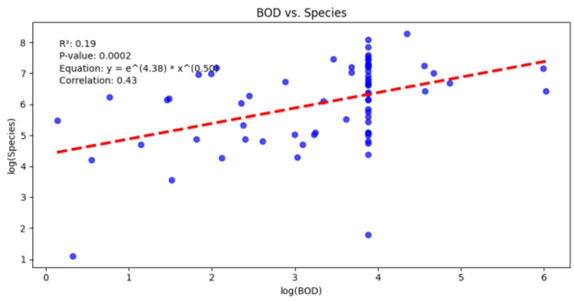


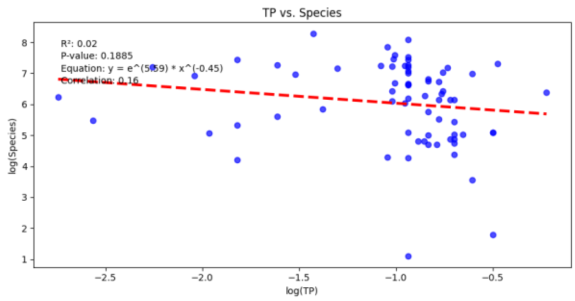


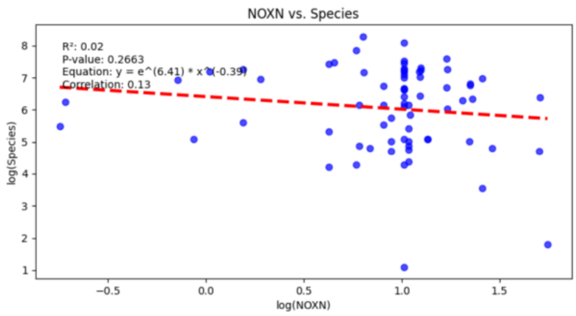












Appendix 6: Power Regression Australia