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SWEDEN AB



ESA DUE DIVERSITY II

Supporting the Convention
on Biological Diversity

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Change Record

Date	Version	Change Record	Authors
01.09.2015	4.0	Final version	Brito, Campos, Gangkofner, Odermatt
11.03.2014	3.0	Quality Assessment Review version	Brito, Campos, Gangkofner, Odermatt
09.12.2013	2.0	Critical Design Review version	Brito, Campos, Gangkofner, Odermatt
21.06.2013	0.1	Preliminary Design Review version	Brito, Campos, Gangkofner, Odermatt

Acronyms and Abbreviations

ANN	Artificial Neural Network
AZE	Alliance for Zero Extinction
BHC	Broken Hill Complex (Australian Bioregion)
BRT	Burt Plain (Australian Bioregion)
CDOM	Coloured Dissolved Organic Matter
CER	Central Ranges (Australian Bioregion)
CHL	Chlorophyll a concentration
CHL FUB	Chlorophyll a concentration retrieved by the FUB algorithm
CHL MPH	Chlorophyll a concentration retrieved by the MPH algorithm
CIBIO-UP	Centro de Investigação em Biodiversidade e Recursos Genéticos, Univ. do Porto
CPUE	Catch per unit effort
EBA	Endemic Bird Area
EGV	Eco-biogeographical Variable
EO	Earth Observation
EU	European Union
FEOW	Freshwater Ecoregions of the World
GAM	Generalised Additive Model
GAW	Gawler (Australian Bioregion)
GBIF	Global Biodiversity Information Facility
GDP	Gross Domestic Product
GIS	Geographical Information System
GLM	Generalised Linear Model
fAPAR	Fraction of Absorbed Photosynthetically Active Radiation
FIN	Finke (Australian Bioregion)
FLB	Flinders Lofty Block (Australian Bioregion)
IBA	Important Bird Area
IBRA	Interim Biogeographic Regionalization for Australia
IGO	Intergovernmental Organization
IOP	Inherent Optical Property
ITCZ	Intertropical Convergence Zone
IUCN	International Union for the Conservation of Nature

LSWT	Lake Surface Water Temperature
LVB	Lake Victoria Basin
LVSWC	Lake Vänern's Society for Water Conservation
MAC	MacDonnell Ranges (Australian Bioregion)
MARS	Multiple Adaptive Regression Splines
MMA	Ministério do Meio Ambiente (Brazilian Ministry of the Environment)
MSAVI	Modified Soil-Adjusted Vegetation Index
NDVI	Normalised Difference Vegetation Index
NGO	Non-governmental Organization
NP	National Park
NPP	Net Primary Production
PQR	Product Quality Report
RF	Breiman and Cutler's Random Forest for classification and regression
RS	Remote Sensing
RUE	Rain Use Efficiency
SD	Secchi Depth
SDM	Species Distribution Model
SM	Soil Moisture
SMUE	Soil Moisture Use Efficiency
SoW	Statement of Work
SSD	Simpson-Strzelecki Dunefields (Australian Bioregion)
STP	Stony Plains (Australian Bioregion)
TSM	Total Suspended Matter
TSS	True Skills Statistics
TRMM	Tropical Rainfall Measuring Mission
UNESCO	United Nations Educational, Scientific and Cultural Organization
WWF	World Wildlife Fund

1 Scope of the Document

This document supports the validation of DIVERSITY II products. A comprehensive literature study and the descriptions of test sites provide an overview of the diversity of conditions where the products of the project will be applied. To assess the quality of the full set of project products, a series of test sites have been defined. These sites will be used to validate project products with respect to the consolidated user requirements. An extensive evaluation of the product quality and representativeness will be performed, through a validation on a representative set of test sites: 10 inland water bodies and 5 drylands. Selection of inland waters sites was based on their representativeness of the diversity in lake water conditions, quality of their lake monitoring programs and, especially, where *in-situ* data for validation are available, while selection of dryland sites was based on representativeness of the drylands biodiversity, distribution amongst WWF ecoregions and inclusion as global biodiversity hotspots. The structure of the document follows the validation scheme that has been developed for DIVERSITY II project.

The document is divided in two main chapters. Inland water test sites detailed description, methodology and results are presented in Chapter 2, while in turn the same is presented for dryland test sites in Chapter 3. Within each main chapter we find for the various test sites: 1) a description on the geographic location, climatic conditions, hydrological and topographical features, geologic background, biodiversity features, important threats to the environment, present levels of protection, and conservation/management practices; 2) a characterization of *in-situ* EO parameters data for bioindicators validation; 3) a description on the available biodiversity data for both status and trend analyses; 4) a report on the validation protocol, including methodology and results on the relationship between EO products and *in-situ* biodiversity data.

The main aim of the validation process is to assess the quality of the full set of Diversity II products. A systematic quality control is crucial because despite the good overall quality of the most recent EO products, some areas can still be found with major errors that could be avoided by an intense review and corroborated with proper validation methodologies. These errors can negatively influence the overall confidence in EO products independently from the accuracy levels of the latter. There are several steps in validation procedures that should be considered (Strand et al. 2007), such as: 1) the distribution of the validation data, which should be generally distributed across the study area in order to cover representative variability; 2) the projections should be equal to avoid biases of the sampling probability of a pixel by its latitude; and 3) experts should interpret the reference data and these should be independent from the experts that produce the outputs to validate. The EO products can be validated through empirical accuracy assessments using ideally *in-situ* data.

2 Inland Waters

To assess the quality of the full set of DIVERSITY II products, a series of test sites have been selected. These sites will be used to validate project products with respect to the consolidated user requirements. An extensive evaluation of the product quality and representativeness will be performed, through a validation on a representative set of selected 30 inland water body sites, of which to an initial set of 10 (Figure 1), a further 20 inland waters were added.

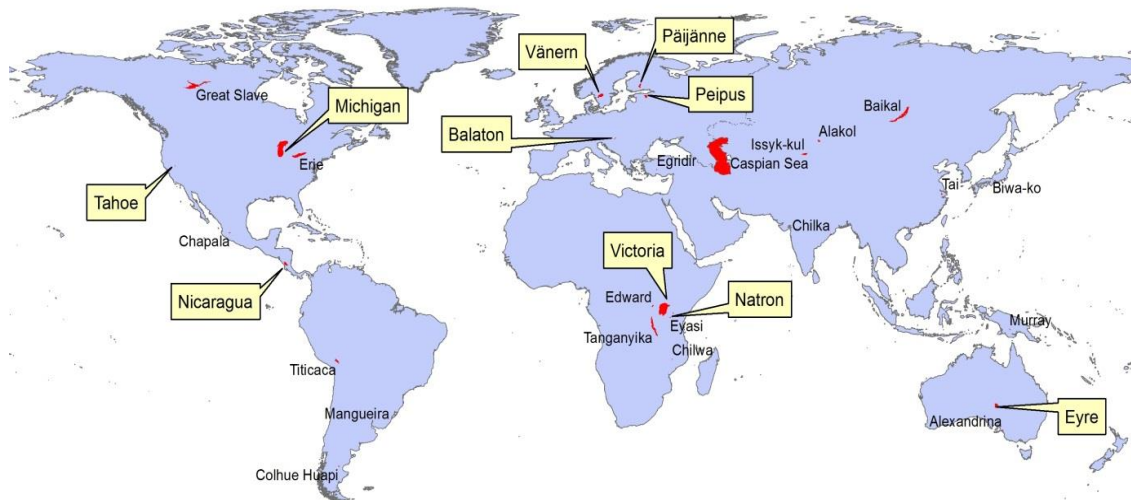


Figure 1 Global map depicting the location of the 30 inland water test sites. Trend analyses will be conducted in the 10 lakes indicated with boxes, while status analyses will be conducted in the full set of 30 lakes.

Selection criteria for test sites were made according to the SoW, which stresses the need to concentrate efforts on biodiversity hotspots and other areas of specific value for the conservation of biodiversity. Additionally, inland water test sites should be: 1) distributed around the world (i.e. most natural lakes and water reservoirs with a size over 500 km² and some natural lakes of biodiversity importance over 100 km²); 2) representative of eutrophic status; and 3) well studied, with available reference data for algorithm development and validation.

Descriptive parameters for each of the 30 inland waters test-sites, which will be used as co-factors in future status analysis, are displayed in Table 1. The list of parameters includes: country and central latitude of the test site, altitude, volume, mean depth, catchment area and population density (based on the gridded population of the world for the year 2000, available at <http://sedac.ciesin.columbia.edu/data/set/gpw-v3-population-density>).

Table 1: Descriptive parameters of the 30 inland water test-sites. Data from LakeNet (www.worldlakes.org) and World Lake Database (<http://wldb.ilec.or.jp/>).

Lake	Country	Latitude (°)	Altitude (m)	Volume (km ³)	Mean Depth (m)	Catchment Area (km ²)	Population Density (pers/5km ²)	Exotic Species (%)
Vanern	Sweden	58.80	44	153.00	27.0	41,182	42.9881	9
Victoria	Uganda, Kenya, Tanzania	-1.50	1,134	2,760.00	40.0	184,000	200.5742	5
Michigan	USA	44.00	177	4,871.00	84.0	117,845	247.9437	33
Balaton	Hungary	46.80	105	1.90	3.3	6,000	144.1735	21
Peipus/Peipsi	Russia/Estonia	58.70	30	25.00	7.0	47,800	35.6585	
Paijanne	Finland	61.50	78	17.80	17.0	25,400	28.4269	24
Nicaragua/Cocibolca	Nicaragua	11.60	32	108.00	13.0	23,844	66.5024	13
Tahoe	USA	39.10	1,897	156.00	313.0	841	90.5207	80
Eyre	Australia	-28.30	-10	30.10	3.1	1,140,000	0.01	
Natron	Tanzania	-2.40	600	0.35	0.5	932	8.8677	
Alakol	Kazakhstan	46.08	347	58.6	22	60,000	4.0857	38
Alexandrina	Australia	-35.43	0.75	1.61	2.8	1,061,469	7.4037	15
Baikal	Russia	53.62	455.5	23,000	740	560,000	5.4282	
Biwa	Japan	35.23	85.6	27.5	41	3,174	751.9714	12
Caspian	Russia, Azerbaijan, Iran, Turkmenistan, Kazakhstan	42	-26.5	78,200	182	3,626,000	96.1665	8

Lake	Country	Latitude (°)	Altitude (m)	Volume (km ³)	Mean Depth (m)	Catchment Area (km ²)	Population Density (pers/5km ²)	Exotic Species (%)
Chapala	Mexico	20.2	1,500	7.9	7.2	53,500	179.7546	14
Chilka	India	19.68	1	4	1.35	3,560	307.1843	
Chilwa	Malawi	-15.33	627	1.8	1	7,500	46.4902	
Colhue Huapi	Argentina	-45.5	258	39.52	17	12,096	0.6445	
Edward	Democratic Republic of the Congo, Uganda	-0.88	912	39.5	17	12,906	84.1023	
Egridir	Turkey	37.87	917	4.38	9	3,309	63.3394	
Erie	USA	41.73	173	458	17.7	78,769	293.1027	28
Eyasi	Tanzania	-3.58	1030		1		40.1472	
Great Slave	Canada	62	156	1,580	41	0.98	0.7861	
Issykkul	Kyrgyzstan	42.92	1,607	1,738	270	15,844	29.5155	50
Mangueira	Brazil	-33.16	8	0.7	2.5		4.1406	
Murray	Papua New Guinea	-7	59		5		1.3971	
Tai	China	31.25	3	4.3	1.9	36,895	571.8627	3
Tanganyika	Tanzania, Democratic Republic of the Congo, Burundi, Zambia	-6.08	773	17,800	572	263,000	57.9046	
Titicaca	Bolivia, Peru	-15.62	3,812	893	107	58,000	60.5897	

2.1 Test Sites

In the following sections, descriptive parameters and broad characterisations of the 10 initial inland water test sites and surrounding areas are presented. Focus is given to the geographic location, climatic conditions, hydrological and topographical features, geologic background, biodiversity features (fauna and flora), important threats to the environment, present levels of protection (Protected Areas), and conservation/management practices in the various regions.

2.1.1 Lake Vanern

Lake Vanern (Figure 2) is the largest lake of Sweden and the third largest in Europe, being located in the southwest of the country, as part of three provinces: Västergötland, Dalsland and Värmland (Willen 1984). Lake Vanern covers a surface area of 5650 km² and has a volume of 153 km³ (mean and maximum depth of 27 and 106m, respectively) (Kvarnäs 2001). Due to a north-south fault zone, the lake is divided into two sub-basins, Värmlandssjön (north) and Dalbosjön (south), which are separated by a shallow archipelago area. In fact, Lake Vanern has many archipelagos with a total of 22.000 islands and skerries (small rocky islands) (Christensen 2002). Also, the lake has more than 100 tributaries, but the River Klarälven that flows into the northern part of the lake is its main tributary, providing up to 35% of the water inflow (Kvarnäs 2001), which makes the drainage basin of Lake Vanern extend over an area of around 47,000 km² up north, reaching even parts of Norway (Olsson et al. 2011). The water residence time is around ten years, and then drained into the Kattegat coastal region (North Sea) via the River Göta älv (largest river in Sweden). Lake Vanern and the River Göta's drainage areas combined cover 10% of the total area of the country (Willen 1984; Kvarnäs 2001; Reinart & Reinhold 2008).

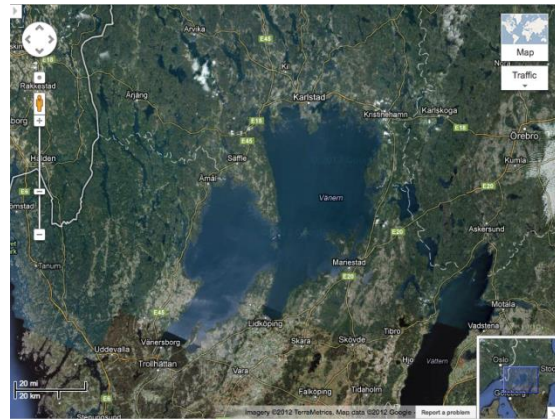


Figure 2: Inland water test site - Lake Vänern.

In terms of water quality, the lake is classified as oligotrophic, with differences amongst the two sub-basins: Värmlandssjön is more humic while Dalbosjön is more turbid. There are two phytoplankton blooms, the one occurring in spring (May-June) being more pronounced than the one in the summer (August) (Strömbeck & Pierson 2001; Alikas & Reinart 2008). All of the above values can vary significantly when considering specific bays and archipelagos, whereof eutrophication and overgrowing algae affect several (MacTaggart et al. 2008).

Lake Vanern and surrounding regions are characterised by a cool-humid climate, with temperatures tending to be more moderate for its latitude due to the influence of the Gulf Stream (mean temperature of the coldest and warmest month being 3°C and 16°C, respectively), and mean annual precipitation around 500 mm (MacTaggart et al. 2008; Kvarnäs 2001; Reinart & Reinhold 2008). Because of these climatic settings, size and shape of the basin,

Lake Vanern rarely freezes up completely, and only shores and areas between archipelagos appear covered in ice during winter (Reinart & Reinhold 2008).

Geologically, the area occupied by Lake Vanern is for the most part consisted of Precambrian granites and gneisses (MacTaggart et al. 2008). In the Last Ice Age, 10,000 years ago, the entire width of Sweden was underwater, connecting the Kattegat (Atlantic Ocean) and the Baltic Sea. Lake Vanern originated when the ice began to melt, which in combination with land uplift due to isostatic rebound, it isolated the body of water as well as the biodiversity within it (Willén 2001; MacTaggart et al. 2008). Nowadays, the lake still harbours many species that are not frequently associated with freshwater habitats, such as the amphipod *Monoporeia affinis*, or glacial relicts like the four-horn sculpin fish (*Myoxocephalus quadricornis*) (Christensen 2002; Piccolo et al. 2012).

Lake Vanern holds many types of habitats, depending whether it is a zone of shallow or deep waters, or if considering the bottom floor, strictly pelagic or coastal biomes (MacTaggart et al. 2008). The shores of the lake are very diverse, including sandy beaches, rocky formations and wet grasslands. The great number of islands and archipelagos further increases the topographic diversity, which makes Lake Vanern a suitable nesting ground for many bird species (Christensen 2002; MacTaggart et al. 2008). These include the marsh harrier (*Circus aeruginosus*), continental cormorant (*Phalacrocorax carbo sinensis*) and sea birds like gulls (*Larus camus* and *L. argentatus*). The lake also harbours endangered bird species, such as the black-throated diver (*Gavia arctica*), turnstone (*Arenaria interpres*), Caspian tern (*Sterna caspia*) and bittern (*Botaurus stellaris*) (Christensen 2002; MacTaggart et al. 2008).

Considering fish fauna, 36 to 38 species can be found in Lake Vanern, the most common being the smelt (*Osmerus eperlanus*), followed by the vendace (*Coregonus albula*). The river lamprey (*Lampetra fluviatilis*), a red-listed endangered species, is also present in the lake (Christensen 2002). However, the most relevant species in terms of fish fauna are the unique landlocked Atlantic salmon (*Salmo salar*) and migratory brown trout (*Salmo trutta*). As explained above, the last direct connection the lake/region had to salt water was to the Baltic Sea, at 10,000 years ago, so the native salmonids (salmon and trout) are more closely related with other salmonid stocks in the Baltic than they are with the Atlantic (Christensen 2002; Säisä et al. 2005; Piccolo et al. 2012). The Vanern salmon represent, along with stocks in two Russian lakes, the only remaining viable, large-bodied, potadromous (fish that require movement through fresh water systems to complete life cycle) stocks in the world (Piccolo et al. 2012). These species have supported very important fisheries since the first human settlements around Lake Vanern. However, in recent times, with the improvement of fishing techniques and increase in catch size, along with the proliferation of hydropower dams in tributary rivers, industrial development and escalation of pollution, the salmonid stocks have suffered reductions in numbers (Piccolo et al. 2012). Currently, only two stocks of wild salmonids are found in Rivers Klarälven and Gullspångsälven, while hatchery production sustains the Lake Vänern's fishery industry (Piccolo et al. 2012).

Besides overfishing of valuable stocks, other major threats to biodiversity (and human health) affect Lake Vanern. Since the 20th century, the lake has been exposed to sewage and wastewaters from industrial and urbanized areas. Paper and pulp mills, as well as a chlor-alkali industry, were responsible for high levels of mercury in the 1960's and 70's, to the point of some fishes at the time not being fitted for human consumption (Christensen 2002). Presently, discharges are low, but mercury still remains accumulated in sediments and fish (Wihlborg et

al. 2004; Wihlborg & Danielsson 2006). Phosphorous levels are considered normal, but nitrogen is still high (Wilander & Persson 2001). Introduced species (non-native) also constitute a serious issue, since they can be responsible for disrupting ecosystem functions, local extinction of species and cause economic impacts (Josefsson & Andersson 2001). These species were brought within ballast tanks or hulls of large ships, introduced to support fisheries, among other means (Josefsson & Andersson 2001). Examples include the Chinese mitten crab (*Eriocheir sinensis*), the zebra mussel (*Dreissena polymorpha*), the American mink (*Mustela vison*), and the Canadian goose (*Branta canadensis*) (Josefsson & Andersson 2001).

Lake Vänern includes two Ramsar sites (Ramsar 2002a,b) the Kilsviken Ramsar site was established in northeast Lake Vänern and in 2001 it was further expanded. Since 2010, the Lake Vänern Archipelago and Mount Kinnekulle were classified as a biosphere reserve by UNESCO, covering an area of 183 km², of which 63% is part of Lake Vänern itself (MacTaggart et al. 2008). Additionally, local societies for water conservation, funded by the Swedish Environmental Protection Agency and Swedish National Board of Fisheries, are promoting actions for the long-term quality of the lake (Christensen 2002).

2.1.2 Lake Victoria

Lake Victoria (Figure 3), situated in East Africa within a shallow depression 1134 m above sea level between the west and east African Rift, is the second largest lake in the world and one of the African Great Lakes, together with Lakes Tanganyika and Malawi (Bootsma & Hecky 2003; Danley et al. 2012). It spans a surface area of approximately 68,800 km², with a convoluted shoreline that stretches by 3500 km, mean depth of 40 m and a water volume of 2760 km³ (Bootsma & Hecky 2003). Lake Victoria is shared

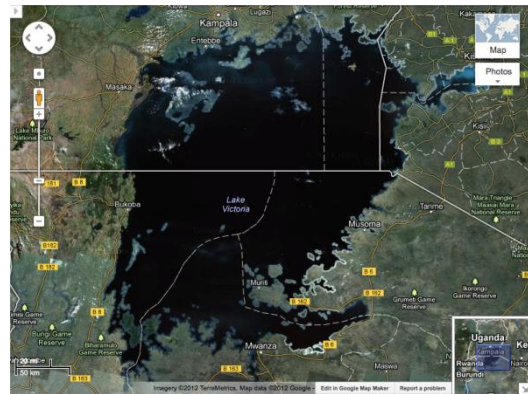


Figure 3: Inland water test site - Lake Victoria.

by three riparian countries: Tanzania (51%), Uganda (43%) and Kenya (6%); but if we consider the Lake Victoria basin (LVB), the drainage area covers up to 195,000 km² and includes also the states of Burundi and Rwanda (Bootsma & Hecky 2003; Kinuthia-Njenga 2008). The lake's main tributaries are the Rivers Kagera and Katonga to the west, water residence time is of 23 years, and then the main outlet is the Victoria Nile at the northern end of the lake (Bootsma & Hecky 2003; Danley et al. 2012). However, the primary water input comes from direct precipitation on the lake surface (80%), while around 85% of the water loss is from evaporation (Danley et al. 2012). Water levels can fluctuate significantly because evaporation is consistent year-round, the same not applying for precipitation (Danley et al. 2012).

Lake Victoria reaches the Equator at the northern end. The climate in the region is influenced by the yearly double passage of the ITCZ (Intertropical Convergence Zone), which results in a prolonged rainy season in March/April, and a second shorter rainfall period in October/November (Burgis & Symoens 1987; Danley et al. 2012). The mean annual temperature of the lake is 25°C and the mean annual precipitation in the region is 1600 mm (Danley et al. 2012). The lake is monomictic, with water mixing occurring during the dry season

between May and August, but in more recent times the lake's water column has become more stable (40% of the bottom water being anoxic) (Danley et al. 2012).

The time of origin of Lake Victoria is still debated, but most authors suggest between 1.6 and 0.40 million years ago, as a result of the uplift of the western arm of the East African rift system that cause a backponding effect and made rivers overspill the basin (Danley et al. 2012). Lake Victoria is a relatively shallow lake because, unlike Lakes Tanganyika and Malawi, it does not lie within a rift basin. Combined with the fact that the water input comes mainly in the form of direct precipitation, Lake Victoria has desiccated completely more than once during times of increased aridity, the most recent of those events happening around 15,000 years ago (Danley et al. 2012).

Despite its recent age, Lake Victoria's biological diversity is very high. Many wetlands and swamps border the lake, supporting many types of vegetation such as *Cyperus papyrus* and *Miscanthidium violaceum* (Chapman et al. 2001). They also sustain numerous waterbirds that include the vulnerable papyrus yellow (*Chloropeta gracilirostris*) and white-winged warblers (*Xenoligea montana*), the locally rare shoebill (*Balaeniceps rex*), as well as other more common species like the great egret (*Ardea alba*) and Reed cormorant (*Microcarbo africanus*) (Baker & Baker 2001; Bennun & Njoroge 2001). Lake Victoria is also home for various amphibians (up to 60 species of frogs in the region), reptiles like the Nile crocodile (*Crocodylus niloticus*), turtles, monitor lizards, pythons, mambas and cobras, as well as mammals such as otters (*Aonyx capensis*, *A. congicus* and *Lutra maculicollis*), hippopotamus (*Hippopotamus amphibious*) and the rare sitatunga (*Tragelaphus spekii*) (Chisara et al. 2001).

But perhaps Lake Victoria is best known for its extremely rich fish fauna, and the extraordinary human-influenced ecological changes that occurred in the last four decades. Until the 1960's the lake was home to more than 500 species, of which the great majority belonged to a species flock of haplochromine cichlids, in addition to cyprinids, catfish, mormyrids and carps (Goldschmidt et al. 1993; Witte et al. 2007). The cichlid fauna in Lake Victoria (also on the other African Great Lakes) is an incredible example of an explosive speciation and adaptive radiation that occurred in very little time (since the last desiccation), which resulted in 200 endemic species that occupied various ecological niches, with each type of habitat containing its own community of cichlids, a phenomena unparalleled to any other group of vertebrates (Goldschmidt et al. 2003). Unfortunately, during the last century commercial fisheries became more and more intensive, human occupancy of the basin greatly expanded, and shorelines and wetlands were transformed mainly for agriculture (Balirwa et al. 2003). Due to overfishing of local target species, exotic species were introduced to support the fisheries, such as the Nile perch (*Lates niloticus*), a large predator, and the Nile tilapia (*Oreochromis niloticus*), a detritivore, with dire consequences (Witte et al. 2007). By the 1980's the Nile perch populations suddenly expanded, over 300 species of haplochromine cichlids of every trophic niche became extinct and food chains in the lake were greatly altered and simplified, with only three species dominating the catchments: the two introduced species and one native cyprinid *Rastrineobola argentea* (Witte et al. 2007; Sitoki et al. 2010; Downing et al. 2012). The introduced Nile perch was responsible for the disappearance of many endemic species, especially phyto- and zooplanktivorous haplochromines (Sitoki et al. 2010). If we add the continuous intensification of human pressures related with agriculture, deforestation and urban development, and the subsequent nutrient enrichment of the lake, the cascading effects were responsible for the eutrophication of the lake (Balirwa et al. 2003; Sitoki et al. 2010,

Fusilli et al. 2013). Water transparency has declined, algae biomass has increased which results in fish kills and deoxygenation of deeper waters, and a water hyacinth *Eichhornia crassipes* increased substantially in numbers and had to be controlled (Sitoki et al. 2010; Fusilli et al. 2013).

In the last two decades, haplochromine cichlids seem to be recovering (as a response to the overfishing of Nile perch) (Balirwa et al. 2003; Witte et al. 2010; Sitoki et al. 2010). It has been suggested that conservation efforts should be more directed to the various satellite lakes that belong to the LVB, since they hold endemic species of their own and relict populations of species that were extirpated in Lake Victoria (Katunzi et al. 2010). Lake Victoria includes two Ramsar sites (Ramsar 2005a,b) and four IBAs (BirdLife International 2013a,b,c,d). Recently, the three riparian states of Tanzania, Uganda and Kenya have established the Lake Victoria Environmental Management Program, which will allow a more coordinate approach to the conservation of the lake's biodiversity (Kinuthia-Njenga 2008). Another good example comes from the definition of action plans to improve urban development in a way that will aid to reduce poverty and improve city waste management through the City Development Strategies scheme (Kinuthia-Njenga 2008).

2.1.3 Lake Nicaragua

Lake Nicaragua (Figure 4), or Cocibolca (indigenous name), is the largest lake in Central America and the main geographic feature of the state of Nicaragua (Lake Nicaragua 2013). At an elevation of 31 m above sea level, it covers a surface area of around 8000 km², with mean depth of 13 m and holding a volume of 108 km³ (Incer 1976; Montenegro-Guillén 2006). Within the lake there are more than 400 islands, many of them volcanic. The main one is Ometepe with an area of 275 km², consisting of two volcanoes, Concepción (active) and Madera (inactive in historical times), linked by an isthmus (Incer 1976). The drainage basin covers a total area of 41,600 km² and includes: the neighbouring Lake Managua and all of its tributaries, which drains into Lake Nicaragua through the River Tipitapa but only periodically during periods of intense rain; the River San Juan basin that is Lake Nicaragua's outlet and follows a 190 km course through the state of Costa Rica into the Caribbean Sea (Incer 1976; Montenegro-Guillén 2006). A mixture of volcanic and tectonic activity during the Early Pleistocene was responsible for the formation of the drainage basin that is located within a geological depression or *graben* (Swain 1966; Incer 1976). Climatically, the region shows a steep gradient with mean annual rainfalls varying between 1200 mm in the northwest, where dry tropical forest and tropical Savannah type vegetation occur, and up to 2000 mm in the southeast, more suitable to rain forests (Incer 1976; Montenegro-Guillén 2006, Portocarrero 2013).

Despite Lake Nicaragua being geographically closer to the Pacific Ocean than to the Caribbean Sea, the biodiversity displayed on the region has more affinity with the latter than to the Pacific



Figure 4: Inland water test site - Lake Nicaragua.

Ocean since Lake Nicaragua is connected with the Caribbean Sea through the San Juan River. The lake harbours more than 40 species of fish, of which the majority belong to a diverse group of cichlids (Thorson 1976, Portocarrero 2013). Similar to the case of the East African Great Lakes cichlids, the Midas cichlid fishes (*Amphilopus citrinellus* species complex) in Nicaragua and Costa Rica have been able to colonize independently various crater lake systems (extinct volcanoes) that occur around the two major lakes, Nicaragua and Managua, and display considerable genetic and phenotypic differentiation (Barluenga et al. 2006; Elmer et al. 2010; Kautt et al. 2012). Another major aspect of the fauna found in Lake Nicaragua is that it holds species most commonly associated with marine ecosystems, such as bull sharks (*Carcharhinus leucas*), sawfish (*Pristis perotteti*), tarpon (*Tarpon atlanticus*), grunt (*Pomadourus spp.*) and snook (*Centropomus parallelus*) (Thorson 1976, Portocarrero 2013).

Threats for biodiversity, as most of the times, are directly related with growth of human presence and development in the lake and surrounding areas, despite the country still showing extreme poverty and having a low technological industrial base (Wickre et al. 2004; Montenegro-Guillén 2006). Mercury contamination due to chlorine-alkali factories and mining activities (especially gold) poses a serious risk for the environment and human health (Wickre et al. 2004; McCrary et al. 2006). The lake is also becoming eutrophic as a result of deforestation, erosion and soil loss, contamination with agrochemicals and pesticides, and uncontrolled urban and industrial waste discharge (Montenegro-Guillén 2006). Native cichlids stocks have declined because of the introduction of African tilapia (*Oreochromis spp.*) during the 1980's, and since then it has had many negative effects (McKaye et al. 1995). Unrelated with anthropogenic pressures (but many times aggravated) is the high vulnerability of the region to natural disasters such as hurricanes, storms, landslides, seismic and volcanic activity which can have dramatic effects on the region's biodiversity (Montenegro-Guillén 2006).

Lake Nicaragua is inserted within the Mesoamerican global biodiversity hotspot (CI 2013a), and is part of two Ramsar sites (Ramsar 2000, 2001) and includes five IBAs (Zolotoff et al. 2009). Although the state of Nicaragua (and Costa Rica) has over 17% of the country covered by protected areas, as well as a good legislative framework to ensure and promote the conservation of biodiversity, there is a clear lack of support and effort to evaluate their impact and to effectively implement these laws (Montenegro-Guillén 2006; McGinley et al. 2009).

2.1.4 Lake Tahoe

Lake Tahoe (Figure 5), located in the north-central Sierra Nevada at an elevation of 1898 m, across the California-Nevada border, is renowned for its deep clear blue waters (Coats 2010). It is the largest alpine lake in North America, with a lake surface of 502 km², and the world's tenth deepest lake (505 m) with a mean depth of 313 m. The volume of the lake reaches 156 km³, and the mean hydraulic residence time is about 700 years (Reutter & Miller 2000; Sahoo et al. 2011). The Lake Tahoe basin is flanked by the Sierra Nevada to the west and the Carson Range to the east, spanning an area of 812 km² that encompasses 63 rivers and

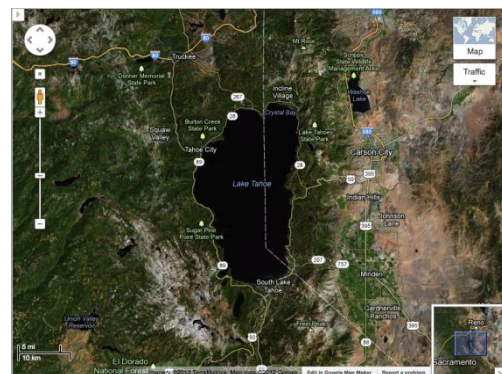


Figure 5: Inland water test site - Lake Tahoe.

streams draining to the lake, with the Upper Truckee River being the largest (Reutter & Miller 2000; USACE 2009). The only outflow of Lake Tahoe is the Lower Truckee River, which passes through the Lake Tahoe Dam and proceeds to flow until reaching Pyramid Lake (USACE 2009). Besides rivers and streams, the drainage basin also includes over 330 lakes, three marshes and two fens, giving the region a relatively high diversity of aquatic ecosystems (Manley & Lind 2005).

The climate is typical Mediterranean, where cool, wet winters alternate with warm, dry summers. About two-thirds of the annual rainfall occurs between November and April, of which 80% are in the form of snow (Manley et al. 2007; Maloney et al. 2011), existing a strong gradient from west to east such that the northeast shore presents around half (660 mm) of the mean annual precipitation of the southwest shore (1440 mm) (Manley et al. 2007). The mean daily minimum temperature during the winter is -6°C and in the summer the mean daily temperatures can exceed 30°C (Manley et al. 2007). The lake is ice-free year around and surface water temperatures commonly range from $4.5\text{--}10^{\circ}\text{C}$ during winter, $18\text{--}21^{\circ}\text{C}$ during summer (USACE 2009). In terms of geologic setting, the Lake Tahoe basin is mainly composed of igneous intrusive rock (typically granitic) and also igneous extrusive rocks (typically andesitic lahar) (USDAFS 2009; Maloney et al. 2011). The geological background, the large volume of the lake and the relatively small drainage basin account for the low nutrient content and low primary productivity, hence the lake's water clarity (Reutter & Miller 2000, USACE 2009, Wittmann et al. 2012). The photic zone can extend to a depth of a 100 m and the entire water column is oxygenated year-round (Wittmann et al. 2012).

Due to the progressive loss of clarity, at a rate of 0.22 m/year, and eutrophication, Lake Tahoe and respective basin has been intensively studied since the 1960's, so biodiversity and its major threats are well documented (Murphy et al. 2000; Sahoo et al. 2011; Tahoe Environmental Research Centre (TERC) 2012). Relative to the fish fauna in the Tahoe Lake basin, there are currently 23 species found, but the great majority are exotic. In the last 150 years, up to 20 species were introduced of which 16 still maintain viable populations (Manley et al. 2000). The region also supports a considerable invertebrate fauna, with up to 810 unique taxa that include various insects and corresponding larvae, oligochaetes, amphipods, ostracods, molluscs, crustaceans, among many others (Manley et al. 2000; Wittman et al. 2012). In terms of flora, the drainage region is home for 110 nonvascular plants (bryophytes) and 1308 species of vascular plants. 612 unique taxa of fungi and lichen are also present (Manley et al. 2000). Remaining vertebrate taxa comprises of: 217 birds, like the bald eagle (*Haliaeetus leucocephalus*), mallard (*Anas platyrhynchos*) and Canada goose (*Branta canadensis*); 59 mammals that include for example the North American beaver (*Castor canadensis*) and river otter (*Lutra canadensis*); six amphibians, such as the long-toed salamander (*Ambystoma macrodactylum*) and mountain yellow-legged frog (*Rana muscosa*); and eight reptiles like the northern alligator lizard (*Elgaria coerulea*) and three species of garter snake (*Thamnophis* spp.) (Manley et al. 2000; Manley & Lind 2005).

The eutrophication of Lake Tahoe is due to past management and exploration of land surrounding the lake, as well as due to the warming trend in response to climate change (Lindström 2000; Coats et al. 2010). Over the past 150 years the region has gone through intense ranching and timber harvesting to the point of very few remnants of virgin forest occurring today (Manley et al. 2000; Manley et al. 2007). Fire suppression also became an issue since it allowed the growth and expansion of species less tolerant to fires. Combined with

more severe droughts, the potential for the outbreak of large and devastating wildfires increased (USDAFS 2009). The Angora Fire, in July 2007, was the largest in 100 years, burning a total area of 12,550 km² (Oliver et al. 2012). All of these issues contribute to soil erosion, which in turn is responsible for the increase in nutrient availability at the lake. This is further exacerbated with the fact that Lake Tahoe is an important tourist site, and there has been considerable degradation and loss of aquatic habitats in the drainage area, such as marshes, for construction of households and businesses (Manley & Lind 2005; USDAFS 2009).

As mentioned before, most of the fish species found at Lake Tahoe are exotic. The only native fish to have gone extirpated is the Lahontan cutthroat trout (*Oncorhynchus clarki henshawi*) and it was because of the introduction of the predators Brook (*Salvelinus fontinalis*) and lake (*Salvelinus namaycush*) trout. Nowadays the only Lahontan cutthroat trout population is managed and is found in one relict stream (Reutter & Miller 2000).

About 85% of the area of the Lake Tahoe basin is managed by the U.S. Department of Agriculture Forest Service Lake Tahoe Basin Management Unit (USACE 2009; Maloney et al. 2011). In the region there are also the Lake Tahoe Nevada State Park, and nine other state parks on the California state side (USACE 2009), allowing a better management of land practices, but at the same time it can become complicated since there are numerous administrative and regulative institutions and agencies (USDAFS 2009).

2.1.5 Lake Michigan

Lake Michigan (Figure 6) together with Lakes Superior, Huron, Erie and Ontario constitute the Great Lakes of North America. Lake Michigan is the second largest of the group by surface area (the fifth largest in the world) and the only entirely located in the United States, surround by four states: Wisconsin, Michigan, Illinois and Indiana (Fuller et al. 1995; Cherkauer & Sinha 2010; Pearsall et al. 2012). It covers a surface area of about 58,000 km², with a shoreline 2640 km long, mean depth of 85 m (maximum of 282 m) and holding a volume of around 4918 km³ (Fuller et al. 1995; GLIN 2013). The drainage basin spans over more than 118,000 km², with the main tributaries being the Fox and the Menominee Rivers in Wisconsin, the St. Joseph, the Kalamazoo and the Grand Rivers in Michigan (GLIN 2013). Water residence time is about 100 years, then Lake Michigan discharges into Lake Huron through the Straits of Mackinac, so in a hydrological sense both constitute only one large lake (the largest in the world in fact) (GLIN 2013). The establishment of the Illinois Waterway (and previous canals and waterways since the late 1800's) allowed the connection of Lake Michigan with the Gulf of Mexico through the Mississippi River (Fuller et al. 2005; GLIN 2013). Lake Michigan is divided into a northern and a southern basin by the Milwaukee Reef, both having a clockwise water flow and small lunar tidal effects have been documented (GLIN 2013). The lake also has a northwestern shallow extension, Green Bay, which its basin represents one-third of the entire Lake Michigan drainage area (Brazner 1997).

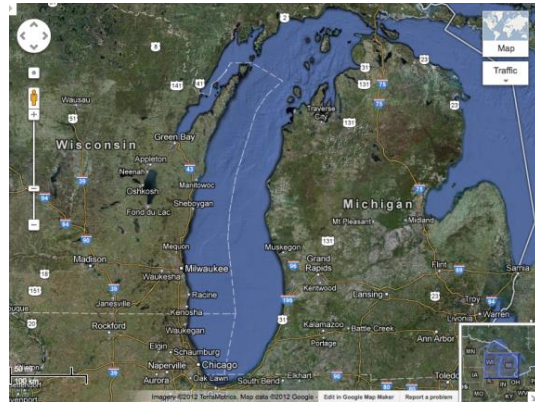


Figure 6: Inland water test site - Lake Michigan.

Lake Michigan's shoreline is very diverse, from wetlands to enormous metropolitan areas like the cities of Chicago and Milwaukee, but perhaps the most important features are the many beaches and the fact that Lake Michigan holds the largest freshwater dune system of the world (Fuller et al. 1995; Pearsall et al. 2012; GLIN 2013).

Climatically, the northern basin has a more cool and dry weather with average annual precipitation around 700 mm and average annual temperature of about 4°C, while the southern basin is a more warmer and humid climate, where the average annual precipitation reaches 1200 mm and average annual temperature 15°C (Fuller et al. 1995). Soil frost and snow cover are common and constitute important aspects of the water and energy cycles of the region (Cherkauer & Sinha 2010). The geological history of the Great Lakes is very recent, being formed 10,000 years ago due to the retreat of the ice sheet during the Last Glacial period, which resulted in large volumes of meltwater and land uplift (Fuller et al. 1995; Pearsall et al. 2012).

Lake Michigan was once home for more than 100 fish species, but many have been extirpated due to overfishing and introduction of exotic species. The fauna includes native species like the lake sturgeon (*Acipenser fulvescens*), lake whitefish (*Coregonus clupeaformis*), muskellunge (*Esox masquinongy*) and lake trout (*Salvelinus namaycush*), as well as many introduced species such as the round goby (*Neogobius malinostomus*), Coho salmon (*Oncorhynchus kisutch*) and Chinook salmon (*O. tshawytscha*), of which some have become invasive (besides being non-native they have threaten biodiversity) like the alewife (*Alosa pseudoharengus*), the sea lamprey (*Petromyzon marinus*) and the zebra mussel (*Dreissena polymorpha*) (Lyons et al. 2000; Brammeier 2001; Bunnell 2012).

The major issues responsible for the degradation of Lake Michigan and its biodiversity are all related to the intense human pressure, which began as early as the arrival of the first European settlers. The northern basin is more forested, human population is sparser and the economy is more dependent on natural resources and tourism, while the southern basin is heavily populated, industrialized and agriculturally rich (Cherkauer & Sinha 2010; GLIN 2013). Nowadays, twelve million people live along Lake Michigan's shores besides the numerous tourists that seek the lakefront (Chicago alone receives 60 million visitors per year) (Brammeier 2001). The lake holds the world's fifth largest oil refinery, 25% of the United States' steel production and the Port of Indiana, the busiest of the Great Lakes (Brammeier 2001). Green Bay and the Fox River are considerably polluted due to a high concentration of pulp and paper mills, some areas being hyper-eutrophic (Brazner 1997). High concentrations of toxic contaminants (including carcinogens) are a problem that Lake Michigan faces as a whole, not only of the water column but also sediments and through the trophic food chain (bioaccumulation and biomagnification). Especially worrying substances are PBDEs (polybrominated diphenyl ethers) (Fuller et al. 1995; Kuo et al. 2010; Evans et al. 2011).

As mentioned before, overfishing had terrible effects on the native fish stocks, especially of lake trout during the 1970's. At the same time, alewives were able to invade the Great Lakes through the Welland Canal to bypass the Niagara Falls and became the most common fish in the lakes. To halt the expansion of this species, other two fishes (Coho and Chinook salmon) were introduced to act as predator of alewives, which in turn also became overabundant (Fuller et al. 1995; Bunnell 2012; DFO 2013). The sea lamprey also reached Lake Michigan thanks to man-made waterways, while the zebra mussel was brought within the

ballast waters of international ships (Brammeier 2001), and throughout the years more and more non-indigenous species (including water plants) are reported (Bunel 2012).

Despite such an intense human presence, Lake Michigan still holds very important national parks along its shore, mainly the Indiana Dunes National Park and the Sleeping Bear Dunes National Lakeshore (Fuller et al. 1995; Brammeier 2001). Many of the stated fish species are listed as of species of concern, threatened or endangered by national and state law (MNFI 2013). Another very relevant aspect to ensure public health and the effective safeguard of Lake Michigan's biodiversity is related with coalition building between the different states, or regional planning that incorporate citizens' concerns and efforts. Examples encompass the Great Lakes Water Quality Agreement signed in 1978 and further reinforced in 1987, or the Lake Michigan Federation founded in 1970, which in 2005 became the Alliance for the Great Lakes (Fuller et al. 1995; Brammeier 2001; Pearsall et al. 2012; AGL 2013).

2.1.6 Lake Balaton

Lake Balaton (Figure 7), in south-west Hungary, is the largest lake in Central Europe. It has an elongated form with a surface area of 593 km² and a shoreline 236 km long. However, the average depth is 3.2 m (maximum depth 12 m) so it only holds a volume of 1.9 km³, making it a large but very shallow lake (Ramsar 2007; ILEC 2013). Lake Balaton can be divided into four basins (from west to east): Keszthely, Szigliget, Szemes and Siófok. The drainage basin covers about 5775 km² that includes 51 watercourses, of which the Zala River is the main tributary (45% of the drainage area), flowing into the smallest (38 km²) and shallowest basin (average depth of 2.3 m), Keszthely. Mean residence time is between 2 and 5 years. The water exits at the Siófok eastern basin, through the only outflow, the Sió Canal that drains into the River Danube. The Kis-Balaton is an important wetland formed by the Zala River, located west of the lake (Juhos & Voros 1998; Istvánovics et al. 2004; Ramsar 2007; ILEC 2013).

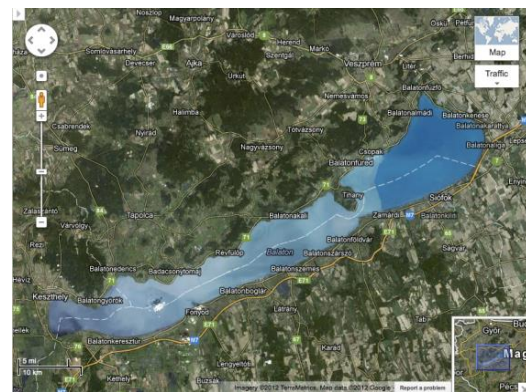


Figure 7: Inland water test site - Lake Balaton.

Lake Balaton is under the influence of a moderately wet continental climate, with average annual precipitation of about 687 mm and mean annual temperature of 10.4°C. During winter the lake freezes, while in the summer the average water temperature is 23°C (Pintér et al. 2008; ILEC 2013). The lake is a young glacial formation that originated 18.000 to 22.000 years ago due to tectonic forces and the retreat of glaciers (Biro 1997; ILEC 2013). The region is dominated by sedimentary rock such as limestone, dolomite and sandstone, so the lake's sediments are highly calcareous (Toth & Fekete 1974; Honti et al. 2007). These physical characteristics make Lake Balaton very susceptible to hydrometeorological forces. Winds are responsible for a permanent mixing of the water column and resuspension of calcareous sediments is high (mean suspended solids around 36 g m⁻³), so the lake is turbid and eu-hypertrophic (eutrophication levels higher in the western basin) (Honti et al. 2007).

Historically, about 2000 species of algae have been identified in Lake Balaton, around 1200 invertebrates and 60 fishes, numbers that have declined due to recent eutrophication

(Bíró 1997; Ramsar 2007). Two relevant examples of algae are *Cladophoraglomerata* and *Bangia atropururea*, while for invertebrates an important species is a endemic Chrysomelidae (water beetle), but there also important benthonic species such as the gravel snail (*Lithoglyphus naticoides*) (Ramsar 2007; ILEC 2013). The lake also harbours many macrophytes (water plants), like *Phragmites australis*, *Lemna minor* and *Potamogeton perfoliatus* (ILEC 2013). In terms of fish fauna, currently the lake holds around 35 species that include stable stocks of sabrefish (*Pelecus cultratus*) and asp (*Aspius aspius*), as well as common bream (*Abramisbrama*), eel (*Anguilla anguilla*), common carp (*Cyprinus carpio*), northern pike (*Esox lucius*), zander (*Stizostedion lucioperca*) and the highly endangered European mud minnow (*Umbra krameri*) (Bíró 1997; Ramsar 2007; Takács et al. 2011). Lake Balaton has also a noteworthy avian fauna, especially migrant and wintering waterbirds, receiving up to 40,000 individuals each season. These include egrets (*Egretta alba* and *E. garzetta*), night heron (*Nycticorax nycticorax*), many duck species (*Anas* spp.), geese (*Anser* spp.), mute swan (*Cygnus olor*) and some raptors like western marsh harrier (*Circusaeruginosus*) and white-tailed eagle (*Haliaeetus albicilla*). Two other vertebrate species to highlight are the European pond turtle (*Emys orbicularis*) and the otter (*Lutra lutra*) (Ramsar 2007).

Quality of Lake Balaton began decreasing since the construction of the Sió Canal in 1860 to better regulate water levels. Early signs of eutrophication were noticed in the 1930's but was during the 1950-60's that a rapid rise in the usage of fertilizers, insecticides and pesticides in surrounding agricultural areas that was responsible for an increase of phosphorous and other nutrient loads in Lake Balaton. The result was several cyanobacteria blooms, massive fish-kills, changes in species composition and abundance. By the 1980's some basins had reached a hypertrophic stage (Bíró 1997; Varkuti et al. 2008). Nutrient load and concentration of pollutants also increased due to intense urbanization and sewage input, especially of the shoreline (Bíró 1997). Lake Balaton has become the most important tourism area of Hungary, receiving more than two million visitors every year (Juhos & Voros 1998). It represents around 30% of the 10% of the tourism contribution to the GDP of Hungary (Varkuti et al. 2008). The region holds 26% of the country's hotels, and about 8900 enterprises are operating in the tourism industry (Pintér et al. 2008).

Lake Balaton has a long historical record of scientific research, making it one of the best-studied lakes in the world (Ramsar 2007; Pintér et al. 2008). Recent implementation of rigorous management practices, such as improved sewer treatment and construction of reservoirs to retain nutrients (mainly from the Zala River), have been able to decrease by 50-60% the phosphorous load and improve considerably the water quality (Varkuti et al. 2008; ILEC 2013). Although there is no official management plan, the quality of the lake is ensured by a series of decrees and acts. Lake Balaton itself is a Ramsar site (Ramsar 2007) as well a IBA (BirdLife International 2013e) and its part of the European Union Habitat and Bird Directives; ILEC 2013).

2.1.7 Lake Peipus

Lake Peipus (Peipsi in Estonian and Chudskoe in Russian; Figure 8) is the fourth largest lake in Europe, as well as the largest transboundary one, located on the border of Estonia and Russia. The lake consists of three unequal basins: Lake Peipus s.s. constitutes the northern, largest and deepest part (area of 2603 km², mean depth 8.3 m, maximum depth 12.9 m,

volume of 22 km³), and it is considered a moderately eutrophic clear water-body; at the south there is the hypertrophic Lake Pihkva/Pskovskoe (708 km², mean depth 3.8 m, maximum depth 5.9 m, volume 2.7 km³); and connecting the latter two basins is the middle strait-like Lake Lämmijärv/Teploe (area 236 km², mean depth 2.5 m, maximum depth 15.3 m, volume 0.6 km³) that is classified close to hypertrophic. The three basins combined cover an area of 3555 km² and hold a volume of 25.1 km³ (Nöges et al. 1996; Nöges 2001; UNDP/GEF 2005; Nöges et al. 2010a; Buhvestova et al. 2011). The drainage basin extends over 44,800 km², most of which belonging to Russia (59%), followed by Estonia (33%), Latvia (8%) and Belarus (0.3%). The main rivers flowing into Lake Peipus are the Velikaya that discharges into Lake Pihkva in the south and drains around 58% of the entire Lake Peipus drainage basin, and the Emajõgi in the west, which discharges into Lake Peipus s.s. and drains 20% of the basin, including the second largest lake of Estonia, Lake Võrtsjärv, that has a surface area of about 270 km². Water residence time in Lake Peipus is around two years, and the only outflow is the River Narva that runs into the Gulf of Finland (Nöges 2001; UNDP/GEF 2005; Mäemets et al. 2010; Nöges et al. 2010b).

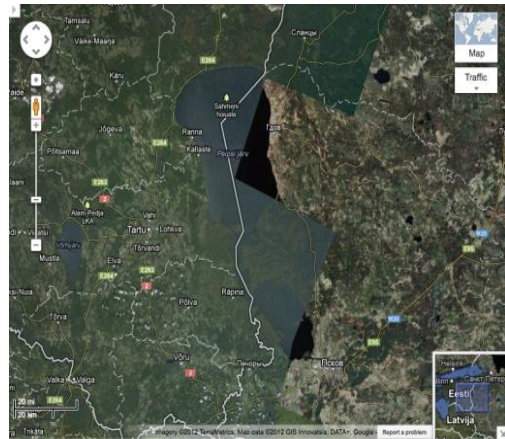


Figure 8: Inland water test site - Lake Peipus.

The Lake Peipus region has a moderate continental climate, with mild winters and warm and wet summers. Mean annual precipitation ranges between 575 and 750 mm, while mean annual temperatures are around 4.6°C. The lake is covered in ice during the months of December to April (Nöges 2001). Lake Peipus was formed around 13,500 years ago due to glacial retreat and isostatic forces (Punning et al. 2008). The bedrock of the basin consists mainly of Devonian sandstone but also some extents of Ordovician/Silurian limestone, which are covered by Quaternary deposits (till, clay and silt) (Punning et al. 2008; Buhvestova et al. 2011). The topography of the surrounding area is predominately flat lowland only slightly undulated, covered mostly by forests, agricultural land, urban areas, peat bogs and wetlands (marshes and swamps) (UNDP/GEF 2005; Roll et al. 2005).

Being large and shallow makes Lake Peipus very susceptible to external factors. Winds are responsible for a constant mix of the water column and sediment resuspension, resulting in an oxygen and nutrient rich environment and only during ice-covered months is when temperature stratification takes place (Puusepp & Punning 2011). Water levels are not regulated, so the lake experiences considerable long-term variations. At historical minimal levels the lake covered an area of 3480 km² and a volume of 20.980 km³, while maximum values reached values of about 4330 km² and 32.128 km³. This means that the surface area may vary by 850 km², water volume by 11.15 km³, and maximum amplitude of water levels is 3 m (Raukas & Tavast 2011).

Lake Peipus holds a very important biodiversity. A large number of plants, up to 122 taxa, can be found. The reed *Phragmites australis* is the most common amongst macrophytes (as a result of eutrophication), but other species can be found such as *Littorella uniflora*, *Potamogeton* spp., and the rare Water awlwort (*Subularia aquatica*) (UNDP/GEF 2005; Kangur et al. 2010; Mäemets et al. 2010). About 500 taxa of macrozoobenthos (macroinvertebrates plus large molluscs) have been recorded (Nöges et al. 1996; Kangur et al. 2010). Lake Peipus

harbours one lamprey and more than 30 species of fish, which include vendace (*Coregonus albula*), bream (*Abramis brama*), perch (*Perca fluviatilis*), zander (*Sander lucioperca*), wels catfish (*Silurus glanis*) and the endemic Peipus whitefish (*Coregonus lavaretus maraenoides*) (Nöges et al. 1996; UNDP/GEF 2005; Bobyrev et al. 2012). The lake and surrounding wetlands are also important areas for birds, like for example black stork (*Ciconia nigra*), aquatic warbler (*Acrocephalus paludicola*), two species of swans (*Cygnus cygnus* and *C. columbianus*), mallard (*Anas platyrhynchos*), smew (*Mergus albellus*), white-tailed eagle (*Haliaeetus albicilla*), osprey (*Pandion haliaetus*), among others (UNDP/GEF 2005).

The main environmental threat in Lake Peipus has been without a doubt eutrophication. It started 40-50 years ago, after the end of World War II. During communist times, intensive but at the same time inefficient use of fertilizers in agricultural practices, as well as poor handling of manure, caused high emissions of nutrients into water-bodies, mainly of nitrogen and phosphorous but also ammonium, other organic matter and ions like sodium, potassium, sulphate and chloride (Roll et al 2005; Mäemets et al. 2010; Nöges et al. 2010b). The concentration of nutrients reached a peak in the late 1980's, and a lake that was classified as mesotrophic during the 1960's was now eutrophic (Roll et al. 2005). The consequences were the increase of volume and frequency of cyanobacteria blooms, expansion of reeds as mentioned above and overall disruption of ecological processes and change of species composition (Nöges 2001; Roll et al. 2005; Mäemets et al. 2010). However, agricultural and industrial practices significantly changed when the Soviet Union collapsed at the beginning of 1990's, causing a reduction of nutrient load (nitrogen and phosphorous loads reduced by 53% and 44%, respectively, between the late 1980's and the mid 1990's) (Roll et al. 2005). So the water quality improved but still, nowadays, the nutrient load is relatively high, mostly carried to Lake Peipus by the two major tributaries that pass through main urban settlements in the region (Nöges et al. 1996; Roll et al. 2005; UNDP/GEF 2005).

Currently Lake Peipus is considered an IBA (BirdLife International 2013f), holds two Ramsar sites as well as two Natura2000 areas (UNDP/GEF 2005). Since the incorporation of Estonia into the European Union (EU), the state has to implement the EU Water Framework Directive. Also many other political and institutional agreements have been established, especially between the two countries that share the lake (Roll et al. 2005).

2.1.8 Lake Päijänne

Lake Päijänne (Figure 9), located in central Finland between the cities of Jyväskylä in the north and Lahti in the south, is the second largest lake of that same state, covering a surface area of 1100 km², and the deepest (maximum depth of 98 m, mean depth of 17 m). The shoreline is 1140 km long and the lake holds a volume of 17.8 km³ (Itkonen et al. 1999; ILEC 2013b). Lake Päijänne can be divided into several sub-basins, with water quality improving towards the south. The northern basin, Ristiselkä, has good water quality but still high levels of

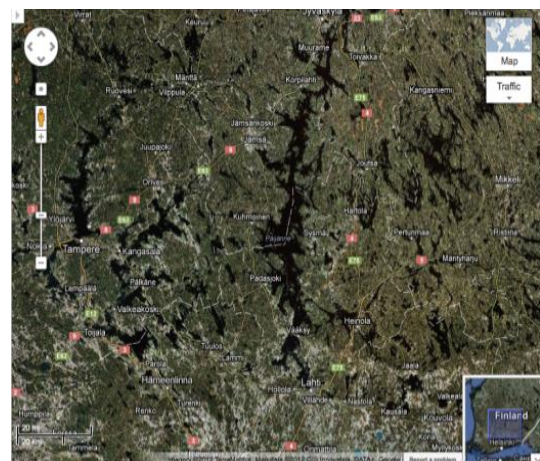


Figure 9: Inland water test site - Lake Päijänne.

nutrients due to historic pollution, while the most southern basin, Asikkalanselkä, as an excellent quality (Itkonen et al. 1999; Partanen & Hellsten 2005; Lusa et al. 2010). The lake has many islands, the largest being Virmailansaari. Lake Päijänne's catchment area is about 25,400 km², the main inflow coming from the Rivers Rutajoki, Muuramenjoki and Arvajanjoki (Syrjänen & Valkeajärvi 2010; ILEC 2013). Water residence time is around 3 years and the lake drains into the Gulf of Finland through the River Kymijoki (ILEC 2013).

The region is subjected to a humid continental climate, where mean annual precipitation is around 576 mm and temperatures about 2.6°C (ILEC 2013). Lake Päijänne was formed approximately 8000 years ago as the result of retreating glaciers and isostatic forces after the last Ice Age, which also reduced the connection to the sea. The geologic background of the lake is predominately Precambrian granites and schist, with sediments consisting mainly of till (ILEC 2013).

Lake Päijänne harbours a typical aquatic fauna and flora of the region. For macrophytes we can find species like *Carex* spp., *Potamogeton* spp., *Sparganium* spp. and *Equisetum fluviatile*, (ILEC 2013). Macroinvertebrates are also diverse and include oligochaetes (*Limnodrilus hoffmeisteri*, *Pelosclex ferox*), bivalves (*Pisidium* spp.), gastropods (*Bithynia tentaculata*, *Valvata piscinalis*), crustaceans (*Asellus aquaticus*, *Monoporeia affinis*) and insects (*Ephemera vulgata*, *Cyrtus trimaculatus*) (ILEC 2013). In terms of larger aquatic fauna, two species of crayfish (*Astacus astacus* and *Pacifastacus leniusculus*), one lamprey (*Lampetra fluviatilis*) and about 31 species fishes can be found in Lake Päijänne, such as *Coregonus* spp. (vendace, whitefish), *Abramis* spp. (common, white and blue bream), *Salmo* spp. (salmon and brown trout), burbot (*Lota lota*), pike (*Esox Lucius*), perch (*Perca fluviatilis*), and many others (Kiskinen et al. 2009b; ILEC 2013).

The water quality of Lake Päijänne has faced very serious eutrophication and pollution problems. Since the 1950-60's until the 1980's, the northern sub-basin of the lake received very high nutrient and toxic loads from pulp, paper and chlorine industries, and sewage effluents, mainly from the city of Jyväskylä (Hattula et al. 1970; Granberg 1992). This resulted in the overall eutrophication of the lake (although the southern sub-basins always maintained relatively good water qualities), changes in species composition and increase of concentration of mercury and chlorinated hydrocarbons in the food chain (Särkkä et al. 1978; Särkkä 1979, 1992; Bagge & Hakkari 1992; Granberg 1992). The settlement of the Water Law at the end of the 1960's, catalysed by the idea of using Lake Päijänne as a source of drinking water for the capital Helsinki, allowed a better control and treatment of sewage waters, and by the 1980's the major industries also decreased their waste discharges into the lake. Nowadays, the northern basins are showing satisfactory results, and around 2/3 of the lake has an excellent water quality (Granberg 1992; Keskinen et al. 2009a). Other ecological issues are related with possible overfishing and lack of connectivity to spawning grounds for salmonids due to the construction of dams (Keskinen et al. 2009a; Syrjänen & Valkeajärvi 2010), and with recreation and tourism, since in the surrounding area live up to 400,000 people (150,000 near the lake) and there are about 30,000 holiday cottages along the shores (Keskinen et al. 2009a; ILEC 2013). In 1993 the Päijänne National Park was established in the southern part of the lake with an area of 14 km² and includes 50 inhabited islands (Keskinen et al. 2009a).

2.1.9 Lake Eyre

Lake Eyre (Figure 10), or Kati-Thanda in the indigenous language, located in central arid Australia, in the northern part of the state of South Australia, is an ephemeral playa (geographic sink) that includes the lowest elevation point in Australia (15 m below sea level), and when it is filled to its maximum capacity (around every two decades) it constitutes the largest lake of the country, with a maximum surface area between 9500 and 10,000 km², mean depth of about 3 m and volume between 15 and 20 km³ (Kotwicki 1986; DeVogel et al. 2004; Tweed et al. 2011; Leon & Cohen 2012). The largest volume (30.1 km³) of the lake ever recorded happened in 1974 (Leon & Cohen 2012). The lake is divided into two sections joined by the Goyder Channel: Lake Eyre North and the much smaller Lake Eyre South (Leon & Cohen 2012).

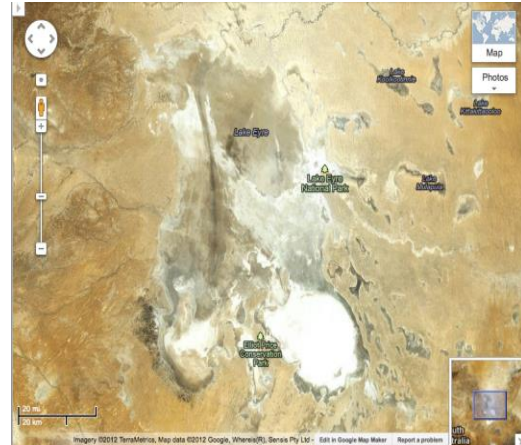


Figure 10: Inland water test site - Lake Eyre.

Lake Eyre is the terminal point of Australia's largest endorheic drainage basin (close inland) that stretches mostly northeast over an area of 1,200,000 km², reaching the monsoon rainfall zone. However the basin is for the great majority within semi-arid and arid zones, with mean annual precipitation ranging between 140 mm in Lake Eyre and 260 mm in the more north-eastern areas (Kotwicki 1986; Tweed et al. 2011; Leon & Cohen 2012). The evaporation rate is extremely high, exceeding 2800 mm (mean annual), so most of the times the region is desiccated, with scattered waterholes separated by large flat plains covered in a 50-cm-thick salt crust in Lake Eyre, and intermittent river flows (Williams 2002; DeVogel et al. 2004; Hesse et al. 2004; Tweed et al. 2011).

The major rivers of the basin, Georgina, Diamantina and Cooper Creek, receive water in their upper catchments from summer monsoon rainfalls, influenced by the El Niño-Southern Oscillation. In years of intense rainfall (La Niña episodes) the water is able to reach Lake Eyre. Inundation of Lake Eyre has recently occurred in 2009 and 2007, while in the past the lake has hold significant water levels in 1949/1950, 1974/1978, 1984/1985, 1985/1986, 1997 and 2000 (Kotwicki 1986; Williams 2002; DeVogel et al. 2004; Costelloe et al. 2005). The flooding of Lake Eyre is largely fed by the Diamantina River, followed by Cooper Creek and Georgina. Other important river systems, which come from the west part of the basin, are the Macumba and the Neales (Webb 2009, 2010; Tweed et al. 2011; Leon & Cohen 2012). The lake then slowly begins to dry out during several months mostly due to evaporation, and water salinity increases to values sometimes higher than in maritime conditions (Williams 2002; Huey et al. 2011). The landscape around the lake is dominated by clay plains and floodplains, as well as dunefields of siliceous sands and stony tablelands (Kotwicki 1986; Tweed et al. 2011). Paleohydrological evidence suggests that Lake Eyre was once a huge waterbody during the late Quaternary, but then in the Holocene it changed from a permanently surface-water system to a groundwater-system (Kotwicki 1986; Huey et al. 2011; Leon & Cohen 2012).

The aquatic biodiversity is relatively low, with species remaining isolated in refugial waterholes during dry periods, and then quickly recolonizing the lake, river beds and floodplains during periods of water flow. While it seems to be very little genetic differentiation

between distinct waterholes of a same catchment basin, there is a high genetic structuring between different river catchments, suggesting that Lake Eyre may be currently acting as a barrier due to its very high salinity. Although riverine species are found in the lake during periods of flooding they do not seem to disperse into adjacent catchments (Williams 2002; Huey et al. 2011). The evidence comes from studies on fishes (Masci 2005; Huey et al. 2008; Faulks et al. 2010) and invertebrates (Carini & Hughes 2004, 2006). The fish fauna includes species like the bony bream (*Nematalosa erebi*), Lake Eyre hardyhead (*Craterocephalus eyresii*), golden perch (*Macquaria ambigua*) and desert goby (*Chlamydogobius eremius*). A vulnerable crustacean, the common yabby (*Cherax destructor*) can also be found at the lake (Huey et al. 2011; ILEC 2013c). When flooded, Lake Eyre constitutes a very important breeding site for many birds, such as the banded stilt (*Cladorhynchus leucocephalus*), Australian pelican (*Pelecanus conspicillatus*), red-necked avocet (*Recurvirostra novaehollandiae*) and silver gull (*Larus novaehollandiae*) (IBA 2013).

Lake Eyre has not been extensively affected by anthropogenic pressures. The human population density is only of 1 person/km², the region has had a long history of occupation by aborigines and pastoral practices have been sustainable even though it includes one of the largest cattle ranches in the world (Tweed et al. 2011; LEB 2013). One issue is related with the conservation of the banded stilt, which is in jeopardy because of large numbers of silver gulls that predate on eggs and chicks of the stilt. The populations of gulls increased as a result of more and more rubbish dumps near urban areas (Williams 2002). Another concern is related with climate change, the effect it will have on future rainfalls, and consequently the possible changes on wildlife cycles. In fact, it may already have been affecting precipitation patterns since Lake Eyre has been more frequently filled in the latter half of the 20th century than in any other time (Williams 2002).

In 1985, the Lake Eyre National Park was established, and the remaining drainage basin holds 37 other national parks and reserves, covering 10.6% of the catchment area, as well as one Ramsar site, the Coongie Lake (Williams 2002; LEB 2013). Lake Eyre is also considered an Important Bird Area (BirdLife International 2013g). Finally, the Lake Eyre Basin Agreement was recognized by the Commonwealth, Queensland and South Australia Governments, to ensure the maintenance of all the values the lake and drainage basin have to offer (Williams 2002; McNeil et al. 2008).

2.1.10 Lake Natron

Lake Natron (Figure 11) is an alkaline-saline (soda) lake best known for hosting the largest population in the world of lesser flamingos (*Phoeniconaias minor*) (WWF 2013j). Located in northern Tanzania, just south from the border with Kenya, at an altitude of 600 m above the sea level, the lake spans 75 km in length and 22-35 km in width, covering a surface area of about 600 km² (can reach up to 900 km²) and being very shallow with depths ranging between 0.5 and 2 m, which allows the lake to only hold a volume of 0.35 km³ (Yanda & Madulu 2005; Bettinetti et al. 2011; WWF 2013j). The region is subjected to a dry climate, with erratic rainfalls below 800 mm per year occurring between December and May, and temperatures many times reaching 40°C or more, exposing the lake to intense evaporation rates (up to 20 mm day⁻¹) that surpasses the water input (Bettinetti et al. 2011; WWF 2013j). Input is mainly in the form of rainfall, besides the lake having four permanent tributaries:

Ewaso Ngiro that drains the Kenyan highlands is the main tributary, but also the rivers Peninj, Moinik and Engare Sero (Yanda & Madulu 2005; Bettinetti et al. 2011). Furthermore, a groundwater reservoir feeds a system of hydrothermal springs, mostly saline or sub-saline, which in turn flow into Lake Natron. Despite all of this, the hydrological balance is still negative, and the lake's only outflow is in the form of evaporation, leaving an evaporite salt crust that it is dissolved during the rainy season (Yanda & Madulu 2005; Bettinetti et al. 2011). The combination of high evaporation



Figure 11: Inland water test site - Lake Natron.

rates and continuous salt input brands Lake Natron the ultimate alkaline lake (pH 9 to 10), with so much dissolved sodium carbonate that it makes the water viscous to touch (WWF 2013j).

The sodium-rich soils and sediments are the result of the intense volcanic activity that occurred during the formation of the Great Rift Valley, and afterwards erosion of volcanic landscape. The lava was alkaline, thus originating highly alkali concentrated waters. Lake Natron was also originated due to the heavy volcanic and seismic activity, which directed water flow into the shallow basin (Yanda & Madulu 2005; WWF 2013j). Since then, Lake Natron has suffered many water level fluctuations, as a response to more arid or humid periods (related to the succeeding glacial cycles), but since 6000-5000 years ago the region has desiccated and the lake has been continuously shrinking to its present size (Yanda & Madulu 2005; WWF 2013j).

Given the physical characteristics of Lake Natron it is not surprising that the biological diversity displayed is low. Still, the lake can harbour astonishing numbers of very few species (WWF 2013j). Blue-green algae (Cyanophyta) dominate the alkaline waters, especially *Spirulina* spp. In turn, these algae support the largest breeding population of lesser flamingos, with up to 500,000 individuals (Woodworth et al. 1997; RSPB 2013; Tebbs et al. 2013; WWF 2013j). However, the flamingos are very sensible to the lake's water levels, and breeding events are irregular, occurring mainly during periods of receding water levels after great floodings (Tebbs et al. 2013). Another species, the greater flamingo (*Phoenicopterus ruber*), can also be found in Lake Natron, as well as other 113 bird species, of which 31 are fowls (Yanda & Madulu 2005; WWF 2013j). In terms of fish fauna, the lake, as well as surrounding streams and springs, is home for alkaline tilapias, two that are endemic (*Alcolapia latilabris* and *A. ndalalani*), and a third one, not endemic, *A. alcalica*. These fishes are able to survive in very warm waters (36 to 40°C) (Bettinetti et al. 2011; WWF 2013j).

The conditions of Lake Natron and surrounding areas can explain the very sparse human populations, with pastoralism by the Maasai still being the dominant land-use in the region (Bettinetti et al. 2011). Nevertheless, the fact that the lake has a potentially high content of soda ash (important raw material for various industrial uses) has made the Tanzanian Government push forward the Soda Ash Project, which could be responsible for a growth of the human population in the area and subsequently increase pollution, disturbance and destruction of the habitat (Yanda & Madulu 2005; RSPB 2013; Tebbs et al. in press). Another threat is the construction of a hydroelectric dam in the Ewaso Ngiro River that will



certainly alter the hydrology and ecology of the lake (Tebbs et al. 2013; WWF 2013j). The lake is also becoming an important tourist attraction, and especially uncontrolled tourism could become an issue (Yanda & Madulu 2005; WWF 2013j).

Unfortunately, Lake Natron is not part of any formal conservation area, and even though it is considered an Important Bird Area (BirdLife International 2013h), and it was established as a Ramsar site in 2001, only in 2009 staff were appointed by the Tanzanian Government to implement the necessary measures (Bettinetti et al. 2011; WWF 2013j).

2.2 Validation of Indicators with Biodiversity Data

Biodiversity distribution data is a fundamental aspect of the validation process of DIVERSITY II products. Such distribution data are essential to determine the accuracy and usefulness of derived biodiversity indicators produced within the scope of the project. As such, distribution data were collected from multiple sources that include: GBIF, Global Biodiversity Information Facility (www.gbif.org); FishBase, Database of fishes (www.fishbase.org); and Distribution Atlases of biodiversity at national or continental level. The Global Biodiversity Information Facility (GBIF) is an international open data infrastructure. It has developed a worldwide information network through which natural history museums and other institutions publish their databases (Guralnick et al 2007). Since its foundation, it has published over 370 million records of species (Costello et al 2013). Despite the amount of data available, there are some concerns about accuracy and quality of the geographic, temporal, and taxonomic coverage (Guralnick et al 2007, Costello et al 2013). Nevertheless, GBIF developed strategies and action plans towards expedited discovery and publishing of primary biodiversity data and for frequent monitoring, and gap analyses (GBIF 2010).

The following chapters describe for both status and trend analyses the available data sources, methodologies used and results.

2.2.1 Status

Available distribution data for inland waters comprises exclusively fish data, the majority of which extracted from GBIF. Although other taxonomic groups would be of interest to the project, such as macro-invertebrates, distribution data for all test sites is unavailable.

The number of fish taxa per lake is represented in Figure 12. Lakes Victoria, Tanganyika, Michigan, Erie, Chilka, as well as the Caspian Sea, are particularly species-rich, holding each more than 100 species. Some lakes however are considerably species-poor, holding each less than 10 species. Low species richness can be related with the natural characteristics of the lake, such as the cases of lakes Eyre, Natron and Eyasi, but it can also be

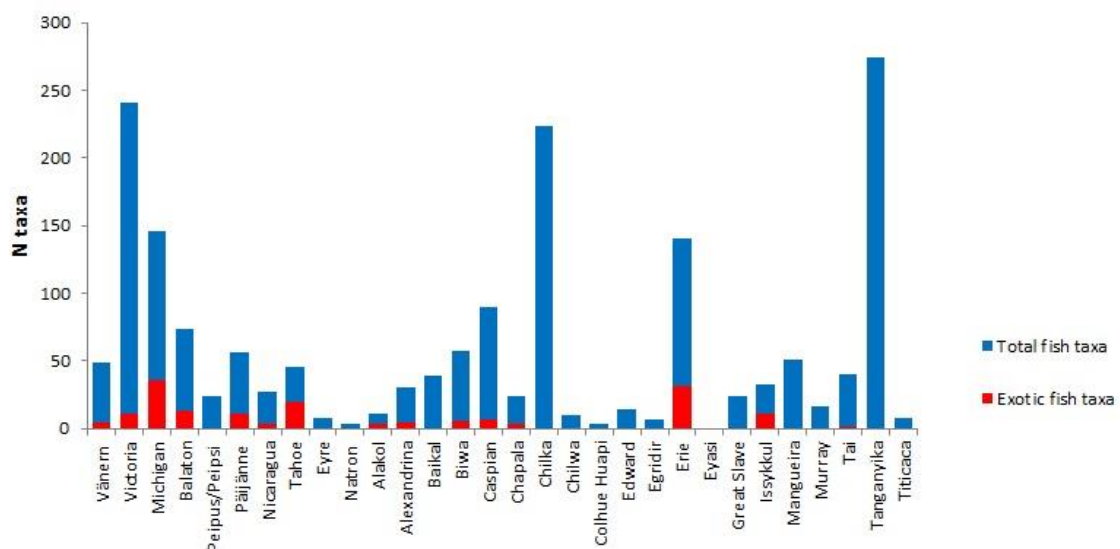


Figure 12: Number of fish taxa and number of exotic fish taxa for the 30 inland water test sites (lakes with no exotic species may also represent lack of available data on exotic species).

explained by the lack of available distribution data, for instance the cases of lakes Colhue Huapi and Egridir.

The estimated number of fish taxa present in each test site may be corrected through the use of local contacts, as well as providing trend data. A list of trend data providers for each of the test sites was compiled (Table 2) and these may work as collaborators and/or end-users of project products.

Table 2: Trend data providers for the initial 10 inland water test-sites.

Lake	Data Provider
Vänern	Petra Philipson - Raw data from LVSWC
Victoria	Fusilli et al. 2013
Michigan	Bunnell 2012
Balaton	Honti et al. 2007
Peipus/ Peipsi	Bobyrev et al. 2012
Päijänne	Syjänen & Valkeajärvi 2010
Nicaragua/ Cocibolca	Portocarrero 2013
Tahoe	Wittman et al. 2012
Eyre	Not available
Natron	Tebbs et al. 2013

Validation of biodiversity indicators for inland waters with *in-situ* biodiversity data will be done at the level of inter-test-site. The aim is to compare species richness present at test sites with biodiversity indicators in order to estimate consistencies of derived products across diverse environmental backgrounds. The validation workflow follows three main steps (Figure 13), which include: collection of biodiversity distribution data and environmental data for each test site; correlating biodiversity indicators with species richness; and correlating species richness with relevant co-factors. These three steps are detailed in the following sections.

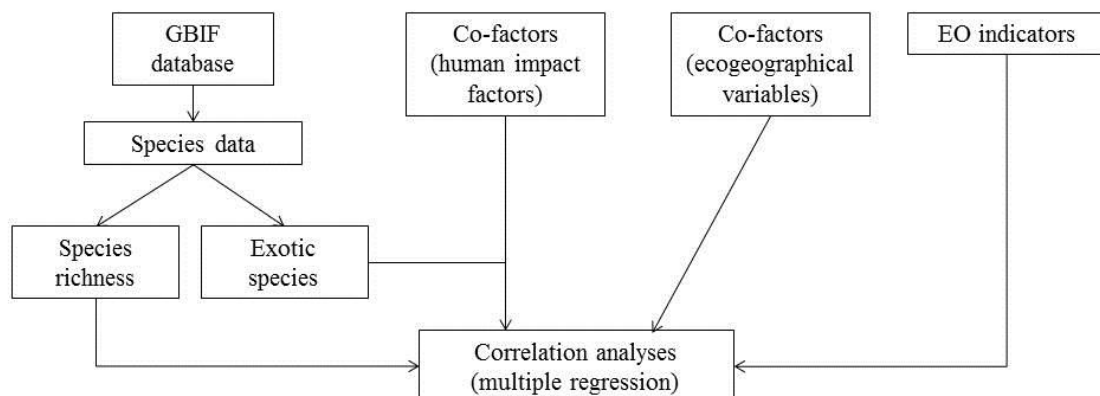


Figure 13: Validation protocol for inland waters based on *in-situ* biodiversity data.

2.2.1.1 Biodiversity and Environmental Data

The following procedures allowed generating taxonomic species lists for each of the test sites (Figure 12). A first set of fish species lists were obtained intersecting the 30 inland water test sites with distribution data available at GBIF, Global Biodiversity Facility Information (www.gbif.org). Quality issues have been raised when using available data at GBIF, namely related with outdated taxonomic identification of species and other taxa, and geographical uncertainties. Because of this, a taxonomic revision was carried out based on the IUCN Red List of Threatened Species (www.iucnredlist.org) and FishBase (<http://www.fishbase.org>). Further taxa were added after checking additional FishBase data and available bibliography, resulting in a final total of 1607 fish taxa (including taxa present in various lakes).

Biodiversity levels present at test-sites should be affected by a multitude of geographical and human-related co-factors (Rathert et al. 1999; Ricklefs 2004; Klop & Prins 2008; Behangana et al. 2009; Stefanescu et al. 2011). As such, seven co-factors likely to influence species richness were compiled for the 30 inland water test sites (Table 1): 1) five geographical factors that include latitude, altitude, volume, mean depth and catchment area; 2) two human related factors, namely population density and proportion of exotic species. Geographical factors were collected from public databases, such as Lake Net (www.worldlakes.org) and World Lake Database (<http://wldb.ilec.or.jp/>), and complemented with available bibliography. Human population density data was extracted from the gridded population of the world for the year 2000, (available at <http://sedac.ciesin.columbia.edu/data/set/gpw-v3-population-density>), while the proportion of exotic species was obtained during taxonomic revision of biodiversity data for each test site.

2.2.1.2 Correlating Biodiversity Indicators with Species Richness and Co-factors

1) Nine year average biodiversity indicators

A first approach consisted of using the nine year biodiversity products derived by DIVERSITY II. These comprise one file per test site, in which data was averaged for the time period between 2003 and 2011. A total of ten biodiversity indicators were available: 1) eutrophication indicators, such as Chlorophyll- α (MPH) (CHL MPH), Chlorophyll- α (FUB) (CHL FUB) and Lake surface water temperature (LSWT d and LSWT n); 2) physical disturbance and contamination indicators that include Total suspended matter (TSM CC), Turbidity (Turbidity CC) and Coloured dissolved organic matter (CDOM FUB); 3) ecosystem indicators that comprise Immersed cyanobacteria, Floating cyanobacteria and Floating vegetation. Mean, minimum, maximum and standard deviation values for the above mentioned indicators were obtained using the "Band Collection Statistics" tool in ArcGIS 10.1.

A total of 47 variables were available for the analysis, which comprised the mean, minimum, maximum and standard deviation of the ten biodiversity indicators, for the period between 2003 and 2011, and also the seven co-factors. Highly correlated variables were discarded, as well as variables with high levels of missing data or very low variability, resulting in a final set of 19 variables. Moreover, inland water sites with extremely high values of species richness (Lakes Chilka, Tanganyika and Victoria all above 200 fish taxa) were excluded as

preliminary analysis showed that the interpretation of the results was hampered by these test sites.

Identification of the most relevant biodiversity indicator, as well as selected co-factors, affecting biodiversity levels at test sites was performed through a multiple regression model (GLZ), using the MuMIn package in R software v. 3.0.2. Models were ranked according to their Akaike's information criterion value. Each model support was estimated through the difference in AICc with respect to the top-ranked model ($\Delta AICc$). AICc rather than AIC is appropriate when there are too many parameters relative to sample size (Burnham & Anderson 2002, 2004). We used the best model to determine the importance of predictors and their significance for each model.

2) Average of monthly products for the nine year period

Some concerns were raised regarding the use of the nine year averaged products, namely the fact that they could be including lake ice periods. In the first attempt, extreme outliers were also noticeable for some of the products (mainly maximum values). So for the second approach, biodiversity indicator data was obtained by averaging monthly products for each test site, between 2003 and 2011. For each test site, monthly products were carefully verified. Monthly products were rejected if they displayed high amounts of missing data (roughly if more than 50% of the lake did not possess data). Missing data was mostly related with noisy components, such as ice and cloud cover, forcing the exclusion of satellite scenes from the analyses. Table 3 indicates the number of months per inland water test site used for the analysis. Lakes Chilka, Tanganyika and Victoria were not included due to their very high levels of species richness, as mentioned above, as well as: 1) Lake Murray, as it was not possible to extract any data from monthly products; 2) Lake Colhue Huapi, which only had 14 months available and preliminary analysis showed it was skewing the results.

Table 3: Number of usable monthly products per inland water test site.

Test Sites	N Months	Test Sites	N Months
Alakol	73	Eyasi	100
Alexandrina	104	Great Slave	40
Baikal	55	Issykkul	104
Balaton	85	Mangueira	101
Biwa	62	Michigan	82
Caspian	101	Nicaragua	49
Chapala	94	Päijänne	55
Chilwa	100	Peipus	60
Colhue Huapi	14	Tahoe	106
Edward	102	Tai	76
Egirdir	107	Titicaca	42
Erie	79	Vanern	72

As in the first approach, we used the same ten biodiversity indicators, and mean, minimum, maximum and sigma values were obtained using the "Band Collection Statistics" tool in ArcGIS 10.1, with the exception of LSWT (d and n) which were extracted using BEAM

5.0. Values were averaged for the nine year period, resulting in a final dataset of 37 variables related with products. Data was then plotted and the coefficient of determination (R^2) was used to assess the fitting of our data to a simple linear regression. Besides using the entire set of test sites (22 lakes), we also divided lakes by their altitude – below (<300 m, with 13 lakes) and above (>300 m, with nine lakes) – and by their latitude – out of the Tropics (17 lakes) and within the Tropics (five lakes) – in order to assess the effects of these two factors.

2.2.1.3 Results

1) Nine year average biodiversity indicators

Multiple regression analysis retrieved nine highly significant indicators that are shown in Table 4.

Table 4: Multiple regression results for inland water status analysis. Indicators and co-factor (Altitude) displayed are the ones most related with the species richness in inland water test sites (GLZ, $p < 0.0001$).

Indicators/Co-factor	β	Std. Error	z	p-value	AICc	ΔAIC	Wi
Altitude	-0.00038	$8.26e^{-5}$	-4.558	$5.16e^{-6}$			
CHL FUB Min	-6.992	0.915	-7.641	$2.16e^{-14}$			
CHL FUB Mean	0.1311	0.01524	8.604	$2.00e^{-16}$			
CDOM FUB Sigma	-12.61	2.317	-5.441	$5.31e^{-8}$	239.49	0	0.73
TSM CC Mean	-0.07182	0.008356	-8.595	$2.00e^{-16}$			
TSM CC Sigma	0.1244	0.01688	7.372	$1.68e^{-13}$			
Turbidity CC Mean	0.01932	0.004451	4.342	$1.41e^{-5}$			
Immersed Cyanobacteria Mean	-3.077	0.4558	-6.752	$1.46e^{-11}$			
LSWT d Sigma	1.988	0.4531	4.387	$1.15e^{-5}$			

Of the best nine models, the two biodiversity indicators with a lowest p-value were CHL FUB Mean and TSM CC Mean (Table 4). Both indicators demonstrate to be negatively correlated with species richness (Figure 14). In general, lakes with higher levels of chlorophyll concentration or total suspended matter are the ones that have lower values of fish species richness, but for both cases species richness is highly variable at low product values. Hence, no strong inferences can be made. These results could be due to the fact that we are combining in the same analysis lakes in various trophic states.

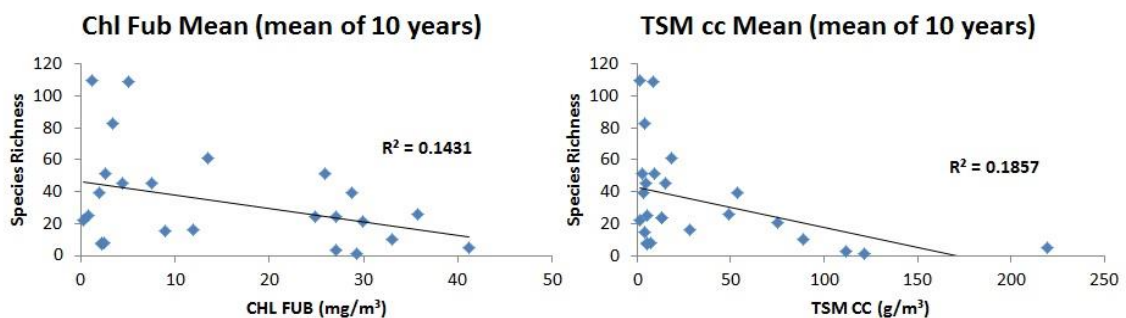


Figure 14: Inland water status data. Scatter plots for the best results, depicting species richness (Y axis) and RS indicators (X axis), per inland water test site (blue diamond shapes).

2) Average of monthly products for the nine-year period

Table 5 summarizes the results for each dataset, highlighting statistically relevant relationships (R^2 above 0.2): Total (Figure 15), below 300 m (Figure 16), above 300 m (Figure 17), outside the tropical area (Figure 18), and within the tropical area (Figure 19 and Figure 20).

All of the ten biodiversity indicators were retrieved at least once in all datasets. For the Total dataset only three positive relationships were observed (with low R^2), namely for Chl MPH Max, LSWT d and n Sigma. Such a few number combined with low R^2 score could, again, be related with the confounding effect caused by mixing lakes of different trophic states, but also geographically diverse (variability in altitude and latitude). Both the number of statistically relevant matches and R^2 score increase in the other datasets.

When considering solely lakes located below 300 m, nine relationships were retrieved, of which seven are negative. The highest R^2 scores were obtained for the Chl FUB Mean and CDOM FUB (Min and Mean). Apparently, in relatively low altitude, higher levels of species richness are found in lakes with lower concentration of chlorophyll and dissolved organic matter (the same happening with immersed cyanobacteria, all of which are related between them). When concentrations increase, the number of fish taxa drops. Surface waters with high chlorophyll levels are normally high in nutrients, such as phosphorus and nitrogen, causing the algae to grow or bloom. After the bloom of algae populations, oxygen levels become depleted, which leads to the eutrophication of inland waters and the extirpation of entire fish communities. Additionally, the dynamics of chlorophyll and phytoplankton in most large rivers of the world is altered by human activities, such as pollutants and cultural eutrophication (Wehr & Descy 1998).

As for higher altitude lakes (above 300 m), seven relationships are highlighted, all related with water temperature (LSWT d and n) except for Floating Vegetation Sigma. The mean and minimum LSWT products are negatively related with species richness (with low R^2 score), while standard variation of water temperature and floating vegetation are positively related. At higher altitudes, water temperature and vegetation are strongly related (Fu & Burgher 2015). In fact, water temperature plays an exceptional role in influencing the biology of inland waters, through determination of thermal habitat distribution and physiological performance of many different species, particularly for ectothermal organisms (the majority of fish), i.e. organisms whose thermal homeostasis is regulated by changes in the environmental temperature (Langan et al. 2001; Arthington et al. 2010). Indeed, water temperature has been referenced as one crucial variable for determining the structure of freshwater fish communities (Crisp 1996). Different fish species tolerate different temperature ranges, and some fish guilds may become dominant over others in certain time periods when temperatures are more suitable. So high variability of water temperature within one water body allows the occurrence of different fish communities throughout a year, and hence contributes to an increase in species richness (Corenblit et al. 2015). However, an extremely high temperature variation, normally due to the continuous climate change, could further threaten species persistence and induce community and ecosystem shifts in water streams (Jaegar et al. 2014; Osborne et al. 2014). These changes may increase water temperature and decrease dissolved oxygen concentrations, which ultimately leads to extreme physicochemical

stress, loss of available habitat and higher competition and predation of native communities by exotic species (Jaegar et al. 2014).

Concerning lakes located out of the tropics, we obtained five statistically relevant matches between species richness and RS products, all of them positive but with low R^2 score. These include the maximum values for Chl MPH, TSM CC and floating cyanobacteria, as well as LSWT d and n Sigma. The number of taxa at higher latitudes (both in the North and South Hemispheres) appears to be greater in lakes that have higher peaks of chlorophyll and suspended matter throughout the nine year period.

The dataset comprising only lakes located within the tropics was the one that retrieved the highest number of relationships (17), but at the same time it only includes five test sites so strong inferences cannot be assured. All products have relatively strong positive relationship with species richness (except Chl MPH Max), even counting two with R^2 above 0.5 (CDOM FUB Sigma and Immersed cyanobacteria mean).

A final mention to the LSWT (both d and n) Sigma, which was the only variable being consistently retrieved in our analysis, besides always showing a positive match with species richness data. So it seems that lakes that display higher variability of water temperature are the ones holding higher number of taxa, independently of their altitude or latitude.

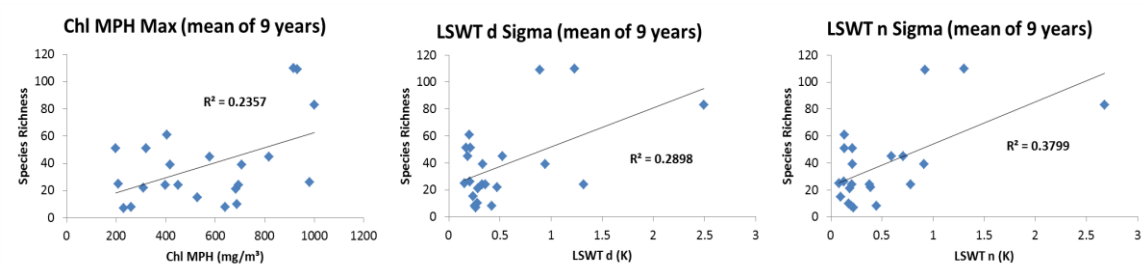


Figure 15: Inland water status data for the Total dataset. Scatter plots for the best results, depicting species richness (Y axis) and RS indicators (X axis), per inland water test site (blue diamond shapes).

Table 5: Summary of statistically significant relationships between species richness and biodiversity indicators, for inland waters. Positive relationships are coloured from light to dark green, while negative are coloured from yellow to red.

Indicators		Total	Altitude		Latitude	
			<300 m	>300m	Out of Tropics	Tropics
CHL MPH	Min					
	Max					
	Mean					
	Sigma					
CHL FUB	Min					
	Max					
	Mean					
	Sigma					
CDOM FUB	Min					
	Max					
	Mean					
	Sigma					
TSM CC	Min					
	Max					
	Mean					
	Sigma					
Turbidity CC	Min					
	Max					
	Mean					
	Sigma					
Immersed Cyanobacteria	Max					
	Mean					
	Sigma					
Floating Cyanobacteria	Max					
	Mean					
	Sigma					
Floating Vegetation	Max					
	Mean					
	Sigma					
LSWT d	Min					
	Max					
	Mean					
	Sigma					
LSWT n	Min					
	Max					
	Mean					
	Sigma					

R ² >	Positive	Negative
0.2		
0.3		
0.4		
0.5		

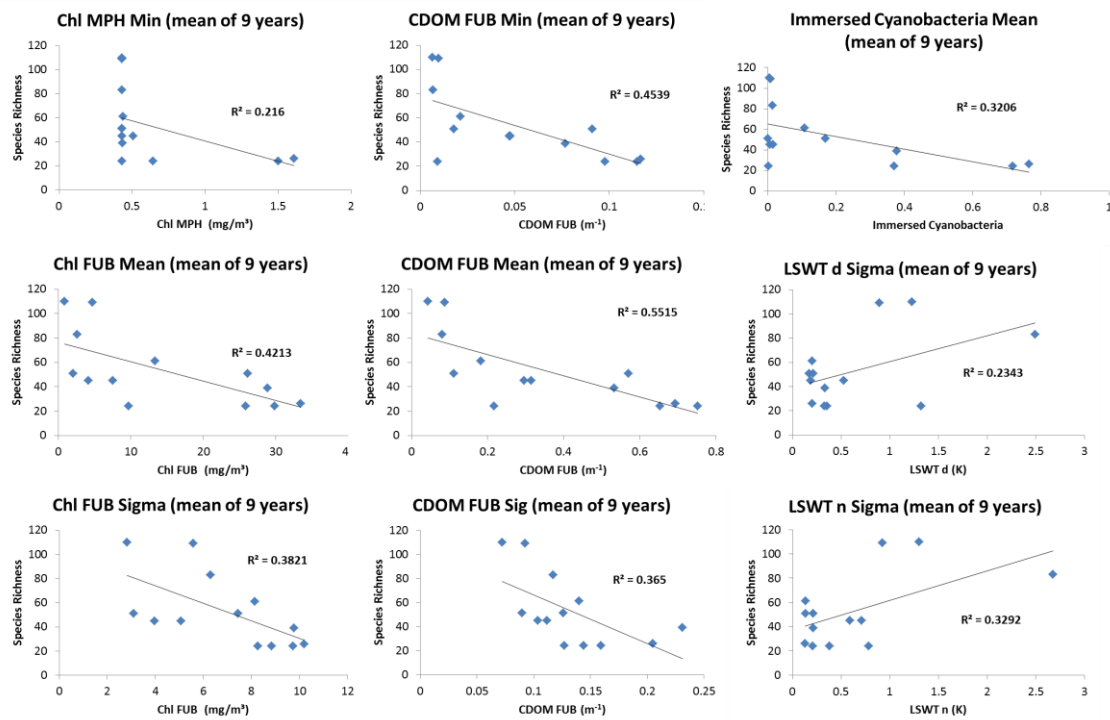


Figure 16: Inland water status data for the <300 m dataset. Scatter plots for the best results, depicting species richness (Y axis) and RS indicators (X axis), per inland water test site (blue diamond shapes).

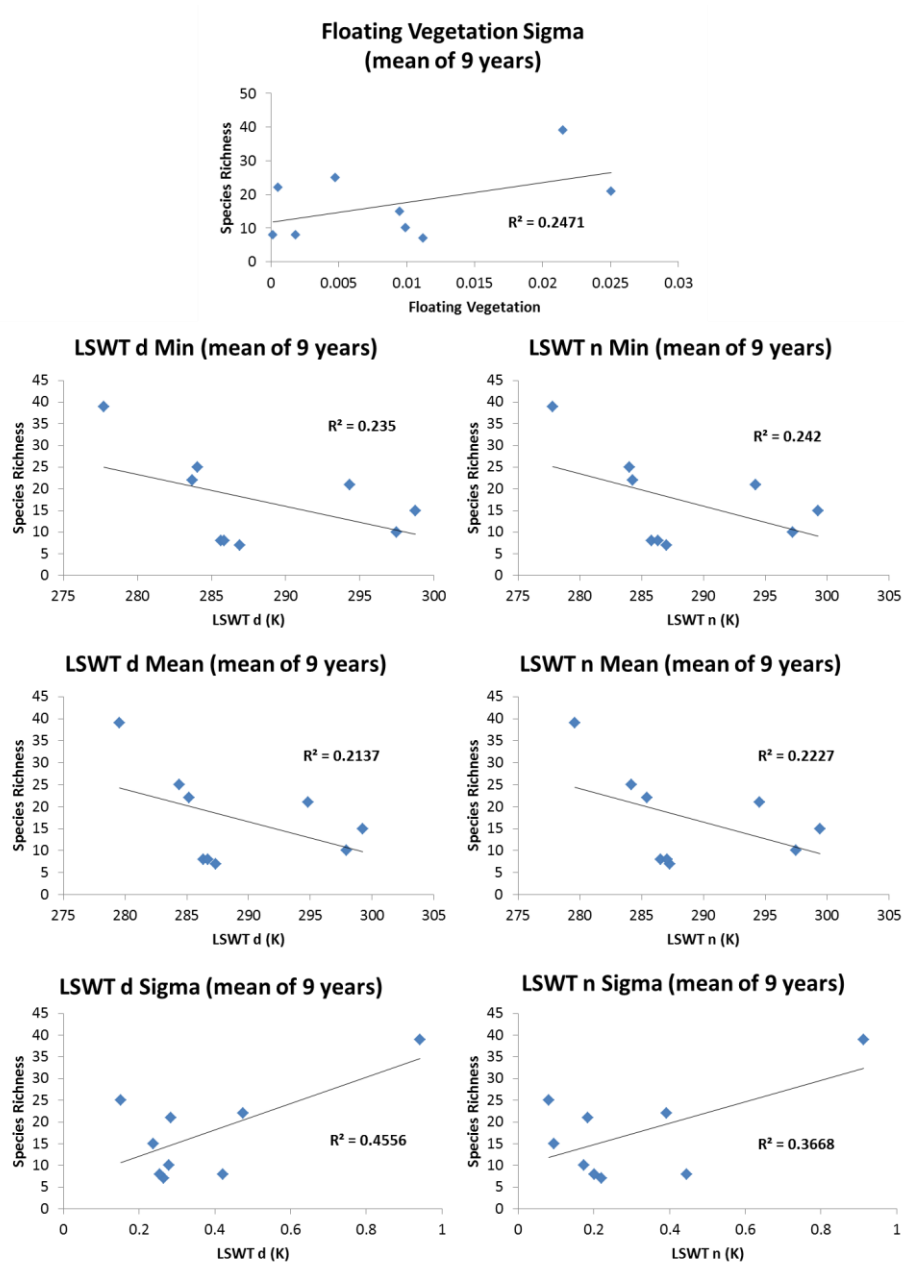


Figure 17: Inland water status data for the >300 m dataset. Scatter plots for the best results, depicting species richness (Y axis) and RS indicators (X axis), per inland water test site (blue diamond shapes).

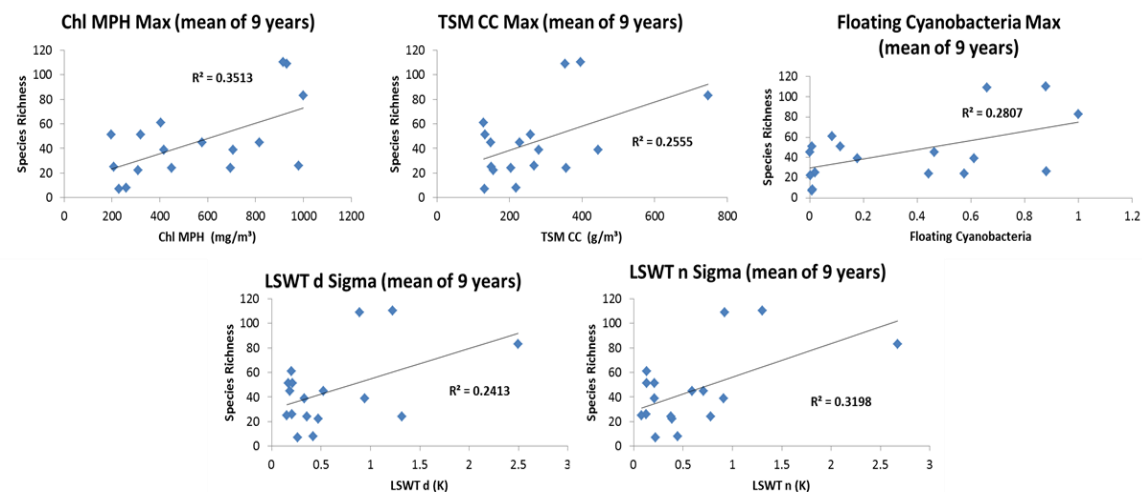


Figure 18: Inland water status data for the Out of Tropics dataset. Scatter plots for the best results, depicting species richness (Y axis) and RS indicators (X axis), per inland water test site (blue diamond shapes).

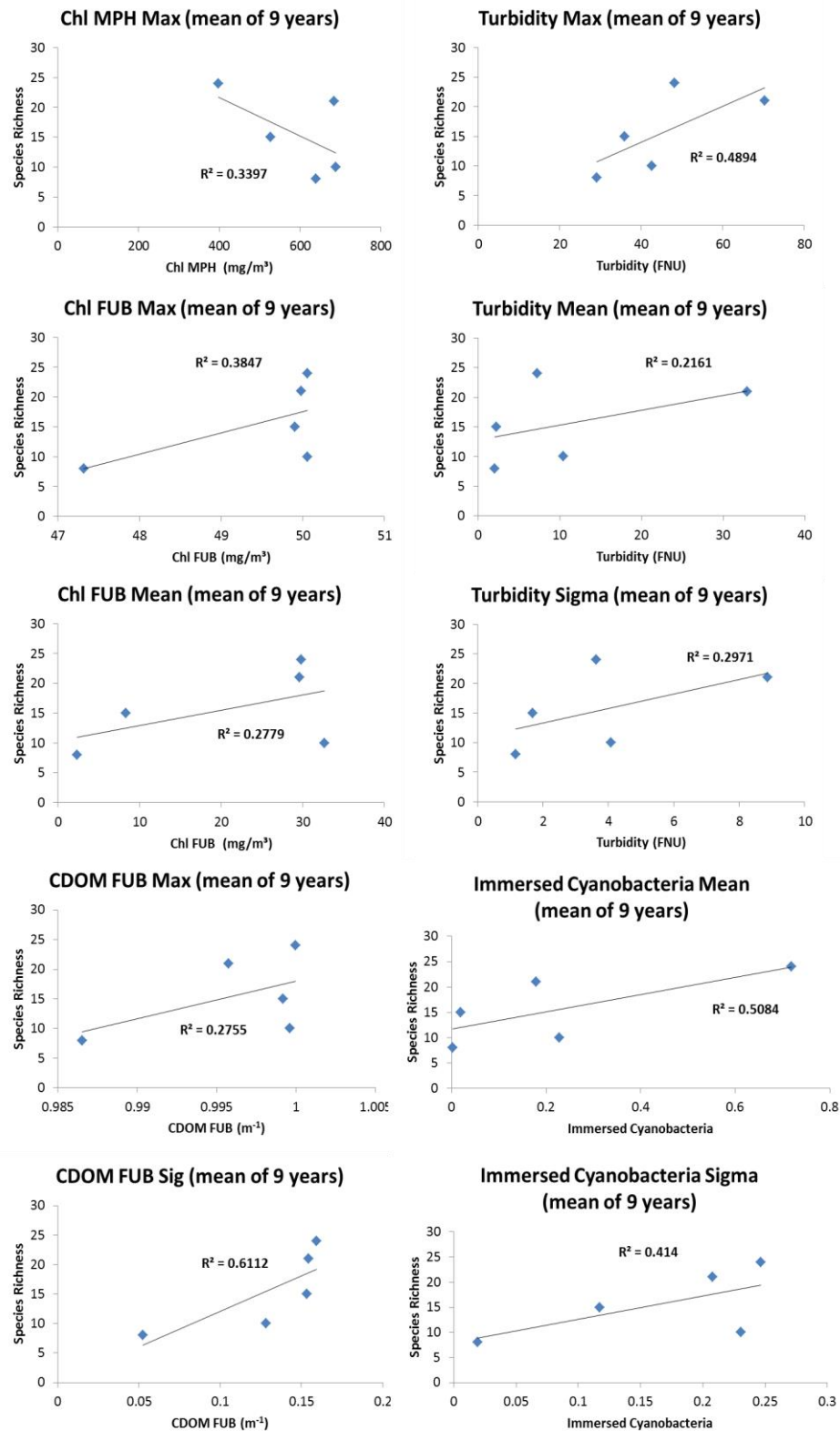


Figure 19: Inland water status data for the Tropics dataset. Scatter plots for the best results, depicting species richness (Y axis) and RS indicators (X axis), per inland water test site (blue diamond shapes).

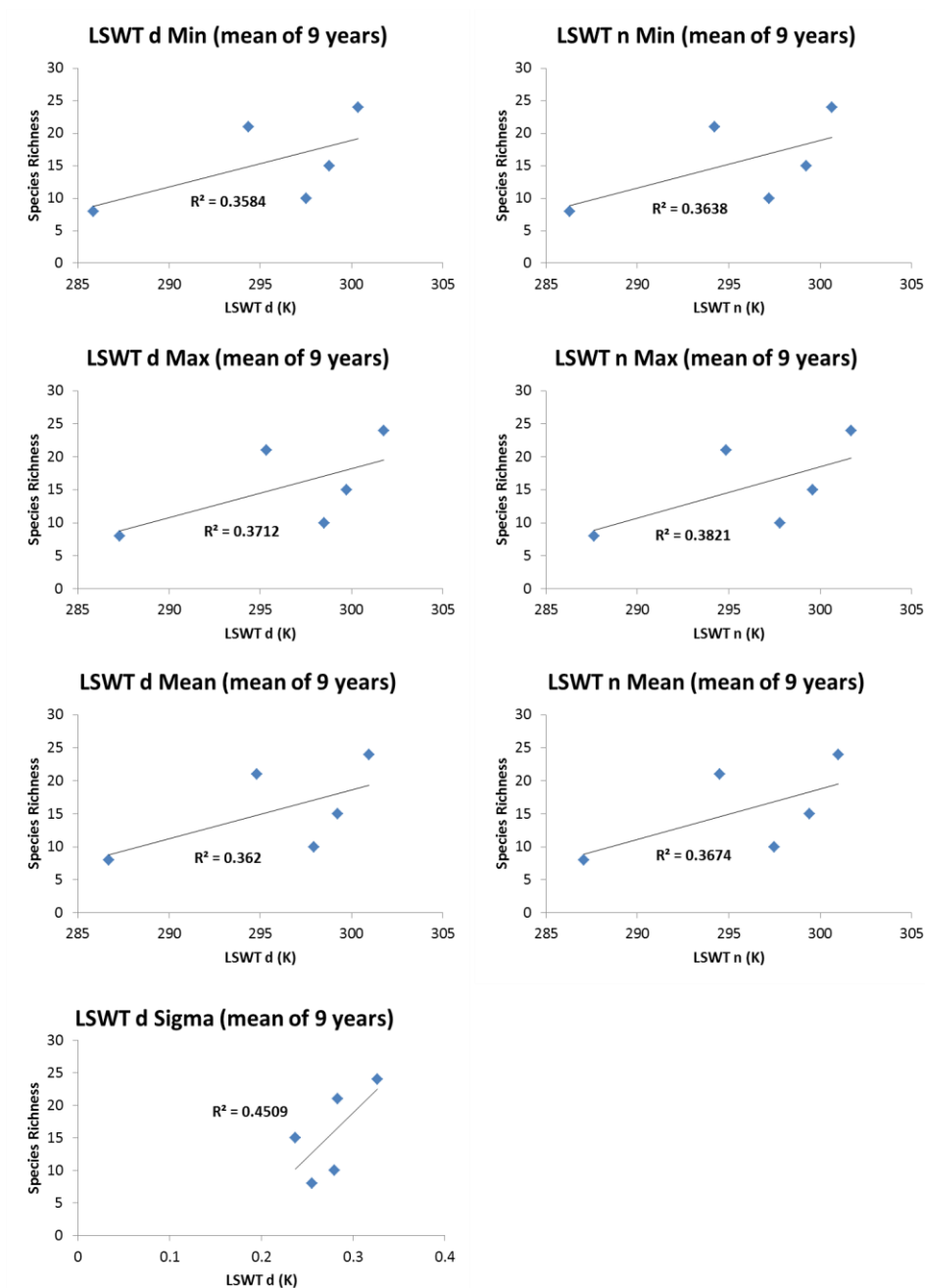


Figure 20: Inland water status data for the Tropics dataset. Scatter plots for the best results, depicting species richness (Y axis) and RS indicators (X axis), per inland water test site (blue diamond shapes).

2.2.2 Trend

Trend analyses have been successfully developed for lakes Vänern, Natron, Michigan, Nicaragua and Päijänne, results being expressed in the following sections. Lakes Balaton and Peipus had very few biodiversity values to conduct any kind of statistical analysis; nevertheless a visual comparison was conducted. For Lakes Tahoe and Victoria no relevant results were discernible due to a lack of correspondence between time series of RS and biodiversity datasets.

2.2.2.1 Lake Vänern

Available biodiversity trend data for Lake Vänern was provided by field experts and comprises yearly test fishing records, namely in 2002 and from 2005 to 2012, which were obtained in 22 field stations distributed throughout the lake (Figure 21). Data consisted in number of counted individuals for captured fish species (Count/net), as well as their weights (Weight/net), both measures then being averaged by the sampling effort (number of nets). Biodiversity indicators were extracted from each monthly product (if sampling was conducted during that month) as an average of 3x3 pixels around the field station centre coordinates.

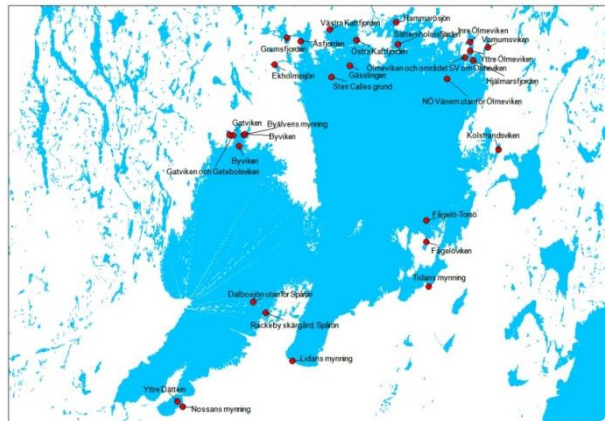


Figure 21: Geographic distribution of the 22 available field stations for Lake Vänern trend data analysis.

Sampling effort throughout the 22 Lake Vänern field stations was uneven, with most stations holding three or less years of fish capture data. As such, we conducted analyses only in the stations with more than three sampling dates (maximum of six dates and only for one station), resulting in a total of six stations. The temporal matches between fish capture dates and biodiversity indicators resulted in three compatible dates to analyse. As an example, Figure 22 illustrates the data available for the station (Fågelövs församling) with best matching possible between biodiversity and indicator measurements.

The approach was then to conduct yearly comparisons between field stations. The idea was to assess if, for example, field stations with high values for an indicator would also be associated with higher numbers of fishes. Preliminary analysis (available in previous PQR versions) showed that stronger correlations were found with Count/net, especially when RS indicator data was extracted for the months of May and June (corresponding to the spring bloom). Lake Vänern is considered an oligotrophic lake (Lake Vänern end-user, personal communication) so we focused our efforts in analysing indicators that are directly related with water quality, such as chlorophyll concentration (for which we used the FUB algorithm instead of MPH, since it is more suitable for this type of water), CDOM, TSM and Turbidity.

Figure 23 illustrates the two best results, which depict a positive correlation between the number of fish (Count/net) and chlorophyll concentration, especially during the month of May. Overall, stations that present higher values of primary productivity (higher

concentrations of chlorophyll a) are the ones that hold also a higher number of fish. The fact that correlations were in general stronger when RS indicator data was taken during the spring bloom is coincidental with the recruitment period for many fish species (Lake Vänern end-user, personal communication).

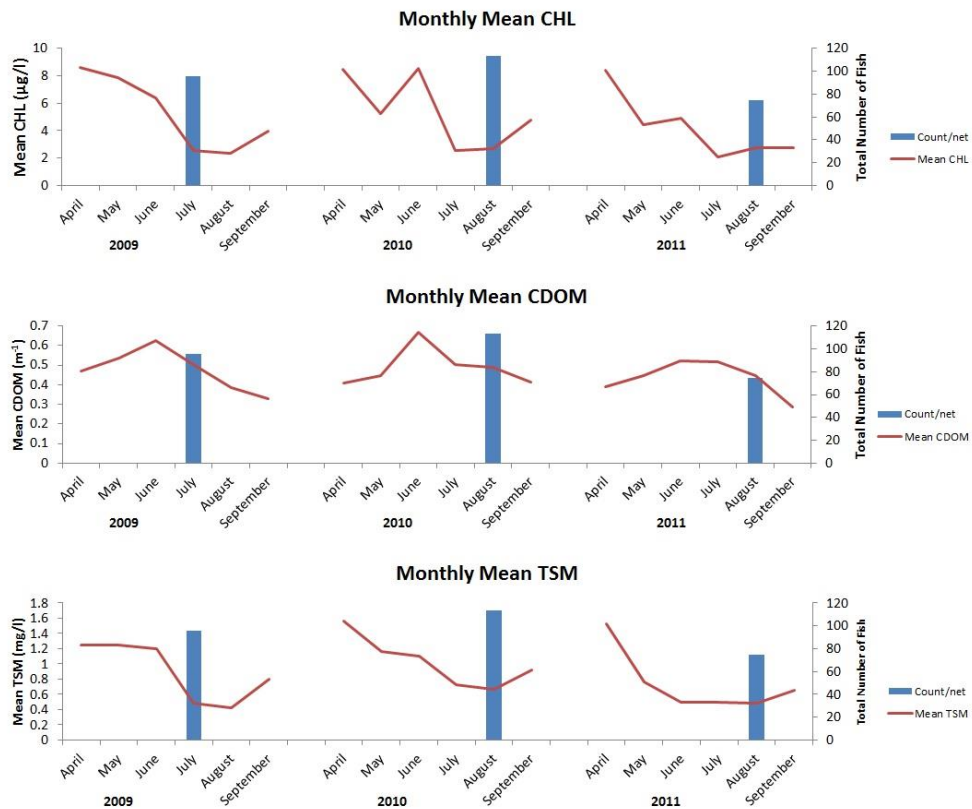


Figure 22: Data available for the Fågelövik station, taking into account the best matching possible between biodiversity (Count/net as an example) and RS indicator measurements. From top to bottom – CHL, CDOM and TSM.

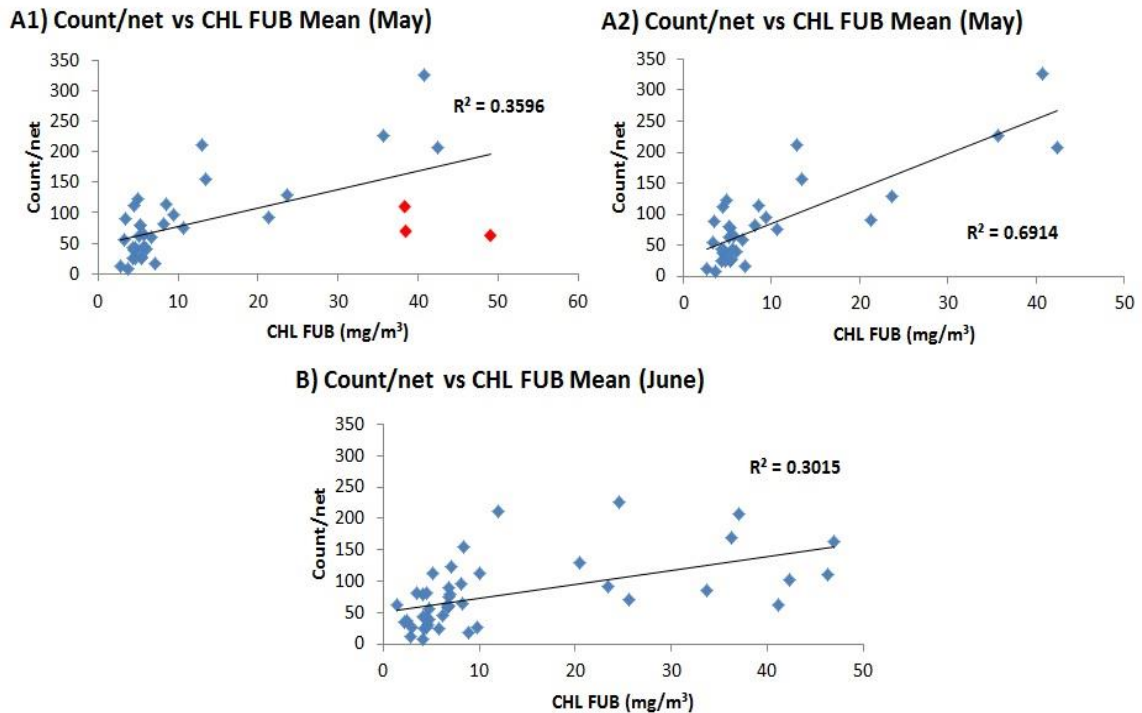


Figure 23: Lake Vänern trend data. Scatter plots for the best results, depicting biodiversity measures (Y axis) and RS indicator (X axis), per field station (blue diamond shapes). A1) Count/net and CHL FUB Mean for the month of May with eutrophic field stations in red; A2) Count/net and CHL FUB Mean for the month of May after removing eutrophic field stations; B) Count/net and CHL FUB Mean for the month of June.

2.2.2.2 Lake Natron

Trend data for Lake Natron was kindly provided by Dr. Emma Tebbs and consisted of breeding event records for the lesser flamingo (*Phoeniconaias minor*) from 2002 to 2011 (Tebbs et al. 2013) (Figure 24).

Analysis of Lake Natron trend data simply consisted of a visual comparison and matching between variation of the extent of the lake (Lake surface area) and the occurrence of lesser flamingo breeding events. Measurements of lake surface area were averaged per month and there were three available parameters: the maximum, corresponding to the sum of all pixels that were at least once identified as water in a given month; the minimum that was the sum of all pixels that were always identified as water in a given month; and uncertainty, which comprised the sum of all pixels for which no valid observations were available in a given month (due to clouds for example).

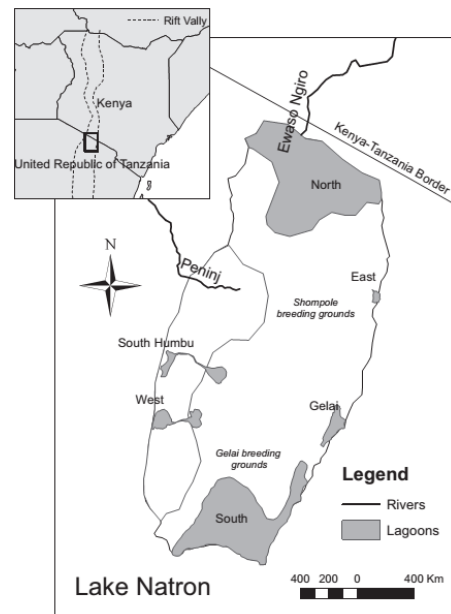


Figure 24: Map of Lake Natron depicting the location of main lagoons and flamingo breeding grounds [adapted from Tebbs et al. (2013)].

The plot of trend data suggests that lesser flamingo breeding events are associated with time periods where the extent of Lake Natron reached a certain peak and then begun to recede (Figure 25). The single exception corresponds to the large breeding event during March-October of 2008 that includes a period where the lake was smaller and then begun to flood. Our results are comparable with those obtained by Tebbs et al. (2013), even though the temporal resolution of our data was not as detailed as by the referred authors.

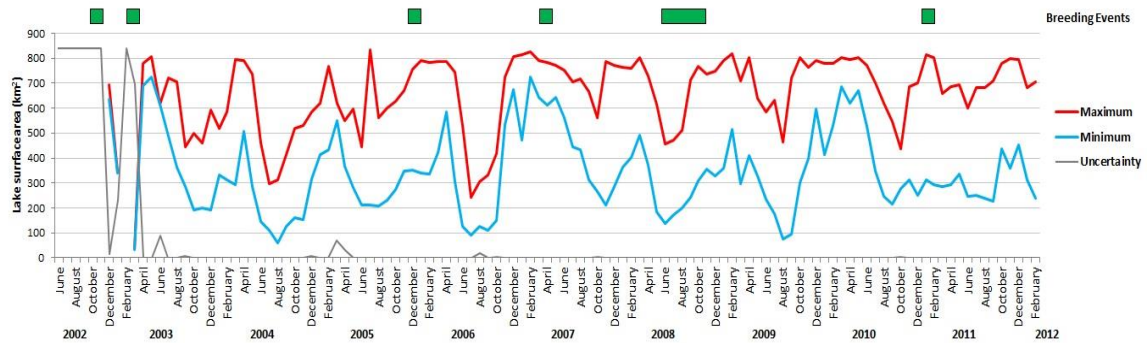


Figure 25: Match between variation of Lake Natron's surface area and lesser flaming breeding events (green boxes) (Tebbs et al. 2013).

2.2.2.3 Lake Michigan

Trend data for Lake Michigan was extracted from Bunnell (2012) and consisted of yearly quantities (from 2002 to 2010) of fishing of seven species – brown trout (*Salmo trutta*), chinook trout (*Oncorhynchus tshawytscha*), coho salmon (*Oncorhynchus kisutch*), lake trout (*Salvelinus namaycush*), rainbow trout (*Oncorhynchus mykiss*), yellow perch (*Perca flavescens*) and sea lamprey (*Petromyzon marinus*). Biodiversity data was measured as yields (millions of kg), except for yellow perch – CPUE (catch-per-unit effort, i.e., number of fish per 305 m of graded-mesh gillnets with equal-length panels), and sea lamprey as abundance. There were ten available RS indicators: CHL FUB, CHL MPH, CDOM FUB, TSM CC, Turbidity CC, immersed cyanobacteria, floating cyanobacteria and vegetation, and LSWT (d and n).

In order to assess if there was any kind of agreement between biodiversity and indicator measurements Spearman's rank order correlation analyses (Spearman's r) were conducted since data did not follow a normal distribution. Spearman's r is simply Pearson's r computed for ranked data (McKillop 2005), almost as statistically powerful as Pearson's r (McGrew & Monroe 1993), and it has been used before in studies assessing relationships between biodiversity and remote sensing data (Ingram et al. 2005; Collin et al. 2011). Analyses were conducted between RS indicators – including the mean, minimum, maximum and standard deviation (sigma) – and each individual species, as well as for all the salmonids together (trout and salmon). Significance of results were tested by their probability values (p), considering a confidence level of $p < 0.05$.

Results of the Spearman analysis are displayed in Table 6. A greater number of statistically relevant correlations were found for the coho salmon, rainbow trout and lake trout. Hence we focused more our attention in those species, yet significant correlations were also found for remaining species (and total salmonids), such as between yellow perch and the standard deviation for floating vegetation.

Figure 26 and Figure 27 display five representative correlations for the coho salmon, rainbow trout and lake trout. Overall, the majority of indicators tend to be negatively correlated with biodiversity data, to the exception of TSM CC and immersed cyanobacteria that show positive correlations.

Table 6: Correlation analysis for Lake Michigan trend data. Correlation values are based on Spearman's r . Values above zero indicate positive correlations, while below zero indicate negative correlations. Probabilities are depicted between brackets for the highest correlation values (≥ 0.5 or ≤ -0.5).

Indicators		Brown Trout	Chinook Trout	Coho Salmon	Rainbow Trout	Lake Trout	Total Salmonids	Yellow Perch	Sea Lamprey
CHL MPH	Max	0.29	0.29	-0.61 (0.106)	-0.63 (0.094)	-0.87 (0.005)	0.21	-0.27	0.2
	Mean	0.31	0.31	-0.63 (0.097)	-0.57 (0.143)	-0.65 (0.081)	0.19	-0.37	-0.04
	Sigma	0.25	0.25	-0.66 (0.073)	-0.63 (0.094)	-0.87 (0.005)	0.17	-0.2	0.12
CHL FUB	Min	0.18	0.18	-0.1	-0.19	-0.11	0.1	-0.51 (0.194)	-0.3
	Max	-0.22	-0.22	-0.55 (0.156)	-0.76 (0.03)	-0.48	-0.31	-0.07	-0.01
	Mean	-0.16	-0.16	-0.75 (0.033)	-0.63 (0.094)	-0.58 (0.133)	-0.21	-0.15	-0.07
	Sigma	0.6 (0.117)	0.6 (0.117)	0.26	0.06	0.04	0.67 (0.071)	0.46	0.73 (0.04)
CDOM FUB	Min	-0.08	-0.08	-0.46	-0.6 (0.114)	-0.64 (0.09)	-0.08	-0.22	0.16
	Max	-0.21	-0.21	0.13	-0.06	0.47	-0.28	-0.11	-0.43
	Mean	-0.11	-0.11	-0.76 (0.028)	-0.63 (0.094)	-0.34	-0.21	-0.49	-0.17
	Sigma	0.57 (0.138)	0.57 (0.138)	0.03	-0.19	-0.16	0.57 (0.136)	0.5 (0.204)	0.5 (0.207)
TSM CC	Min	-0.32	-0.32	-0.09	0.06	0.08	-0.33	0.49	-0.19
	Max	-0.19	-0.19	0.63 (0.097)	0.63 (0.094)	0.83 (0.011)	-0.12	0.15	-0.05
	Mean	-0.49	-0.49	-0.33	-0.13	-0.39	-0.48	0.12	-0.3
	Sigma	-0.41	-0.41	0.5 (0.204)	0.63 (0.094)	0.48	-0.26	0.22	0.04
Turbidity CC	Min	0.38	0.38	-0.18	0.06	-0.1	0.38	0.24	0.43
	Mean	-0.29	-0.29	-0.6 (0.115)	-0.69 (0.057)	-0.69 (0.06)	-0.36	0.02	-0.35
	Sigma	-0.49	-0.49	-0.53 (0.179)	-0.69 (0.057)	-0.63 (0.097)	-0.57 (0.139)	-0.15	-0.43
Immersed Cyanobacteria	Mean	-0.18	-0.18	0.65 (0.081)	0.57 (0.143)	0.87 (0.005)	-0.1	0.07	-0.06
	Sigma	-0.12	-0.12	0.52 (0.191)	0.44	0.82 (0.013)	-0.05	0.2	-0.1
Floating Cyanobacteria	Sigma	0.13	0.13	0.14	-0.06	-0.36	0.17	0.81 (0.016)	0.25
Floating Vegetation	Sigma	0.05	0.05	-0.06	0.03	-0.5 (0.204)	0.14	0.34	0.39

Indicators		Brown Trout	Chinook Trout	Coho Salmon	Rainbow Trout	Lake Trout	Total Salmonids	Yellow Perch	Sea Lamprey
LSWT d	Min	0.01	0.01	-0.79 (0.021)	-0.69 (0.057)	-0.87 (0.005)	-0.12	-0.22	0.01
	Max	0.04	0.04	-0.75 (0.033)	-0.69 (0.057)	-0.92 (0.001)	-0.07	-0.07	0.11
	Mean	-0.13	-0.13	-0.79 (0.021)	-0.69 (0.057)	-0.81 (0.015)	-0.26	-0.22	-0.06
	Sigma	0.32	0.32	0.61 (0.106)	0.44	0.57 (0.143)	0.4	0.29	0.12
LSWT n	Min	-0.13	-0.13	-0.85 (0.008)	-0.82 (0.013)	-0.86 (0.007)	-0.29	-0.27	-0.11
	Max	0.04	0.04	-0.22	-0.31	-0.63 (0.097)	0.02	0.59 (0.127)	0.05
	Mean	-0.13	-0.13	-0.79 (0.021)	-0.69 (0.057)	-0.81 (0.016)	-0.26	-0.22	-0.06
	Sigma	0.35	0.35	0.28	0.38	0.2	0.4	0.41	0.06

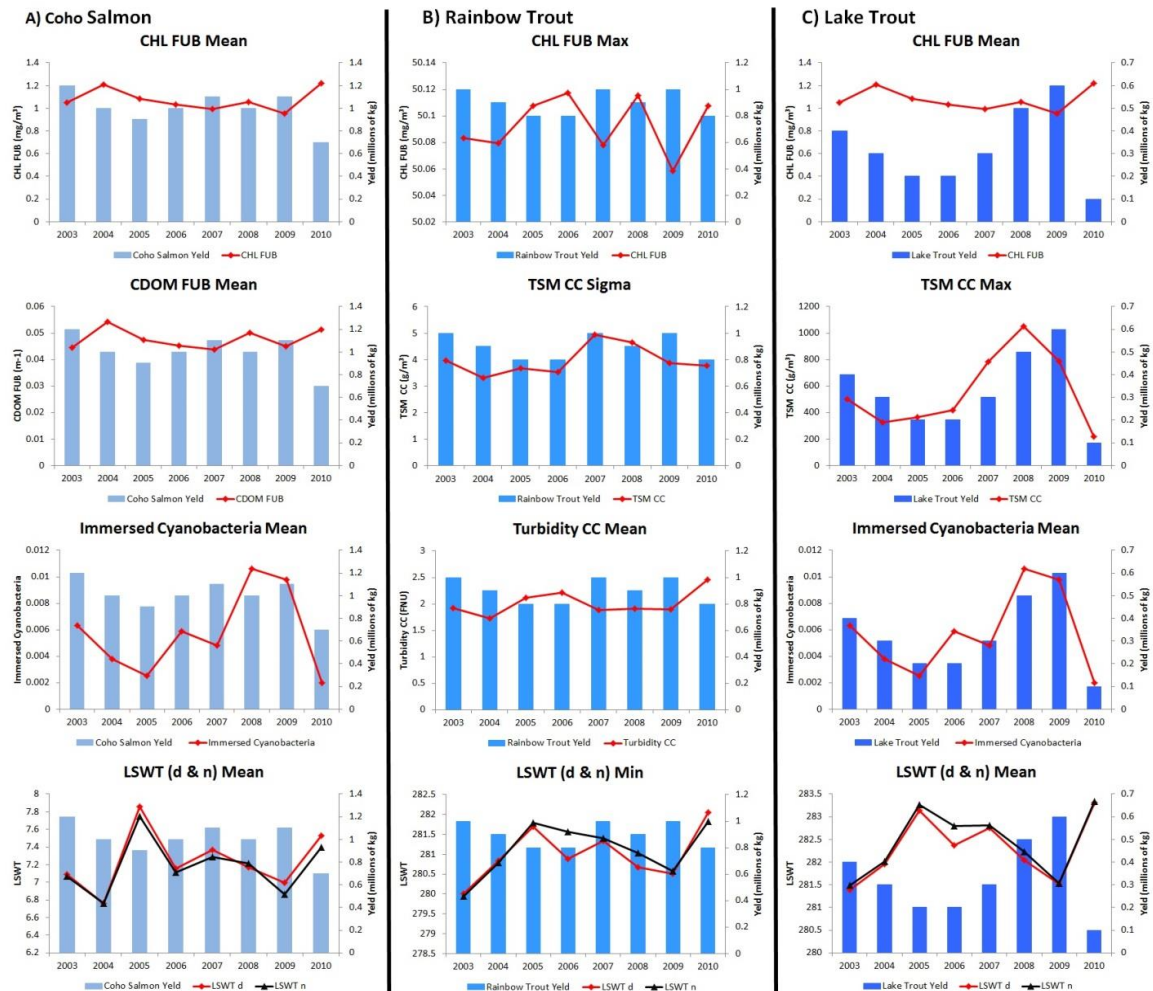


Figure 26: Yearly fish captures of three species and RS indicators for Lake Michigan trend analysis (Bunnell 2012). Charts depict biodiversity measure (right Y axis, blue bars) and RS indicator (left Y axis, red and black lines), from 2003 to 2010 (X axis). A) Coho salmon yearly yield depicted, from top to bottom, with CHL FUB Mean, CDOM FUB Mean, Immersed cyanobacteria Mean and LSWT (d & n) Means; B) Rainbow trout yearly yield depicted, from top to bottom, with CHL FUB Max, TSM CC Sigma, Turbidity CC Mean and LSWT (d & n) Means; C) Lake trout yearly yield depicted, from top to bottom, with CHL FUB Mean, TSM CC Max, Immersed cyanobacteria Mean and LSWT (d & n) Means.

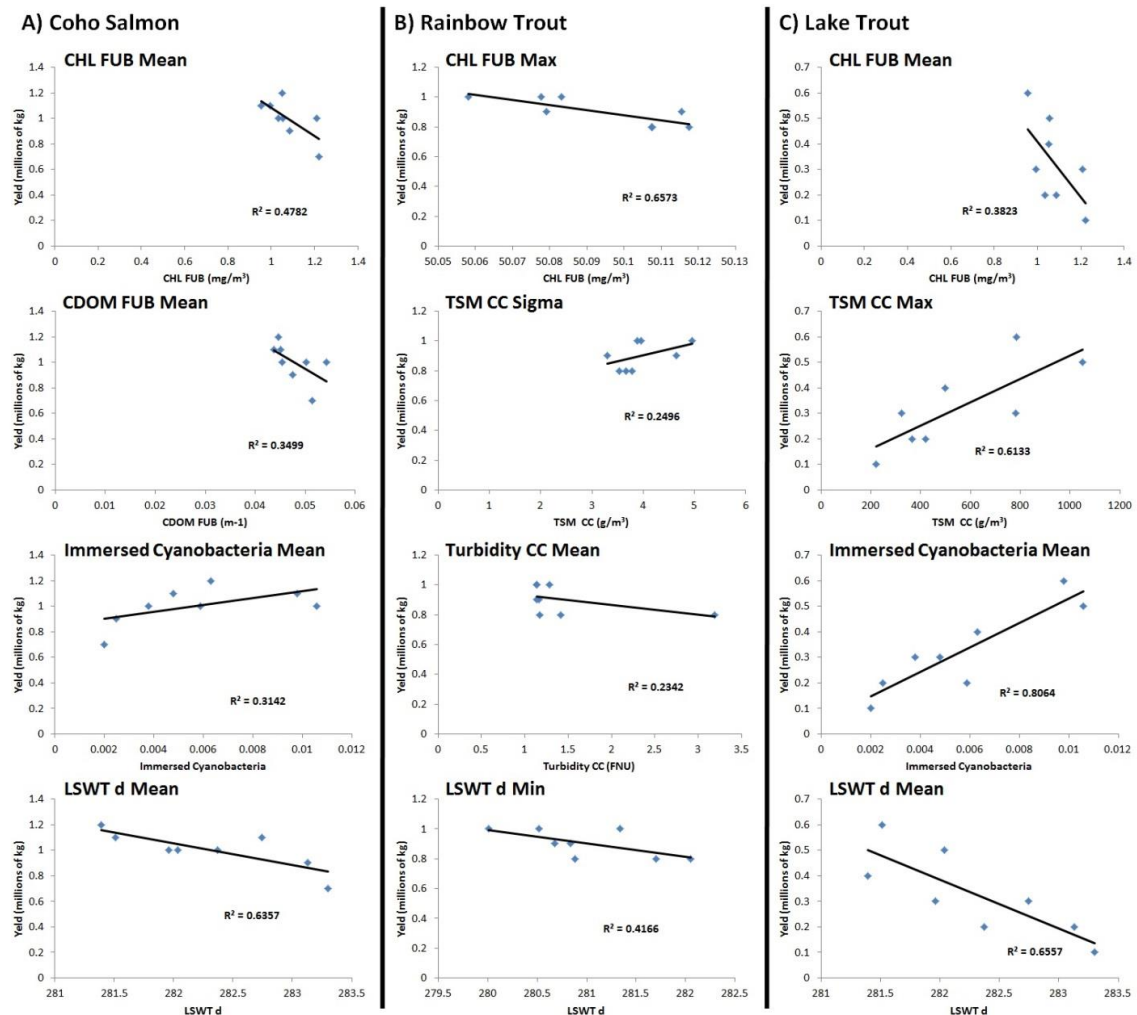


Figure 27: Statistically relevant correlations between indicators and biodiversity data based on Spearman's r . Scatter plots depict fish capture data (Y axis, yield) and RS indicators (X axis), per year (blue diamond shapes). A) Coho salmon yearly yield depicted, from top to bottom, with CHL FUB Mean, CDOM FUB Mean, Immersed cyanobacteria Mean and LSWT (d & n) Means; B) Rainbow trout yearly yield depicted, from top to bottom, with CHL FUB Max, TSM CC Sigma, Turbidity CC Mean and LSWT (d & n) Means; C) Lake trout yearly yield depicted, from top to bottom, with CHL FUB Mean, TSM CC Max, Immersed cyanobacteria Mean and LSWT (d & n) Means.

2.2.2.4 Lake Balaton

Trend data for Lake Balaton was extracted from two sources. First, Tátrai et al. (2008) included yearly (2003, 2004, 2006 and 2007) fish abundance and biomass for the entire lake, as well as three subdivisions (eastern, middle and western basins). The second source, Balogh et al. (2008), comprised monthly data on zebra mussel biomass for four sampling sites, as well as monthly data on total average number of water birds for the entire lake. Both types of data were collected from 2003 to 2005.

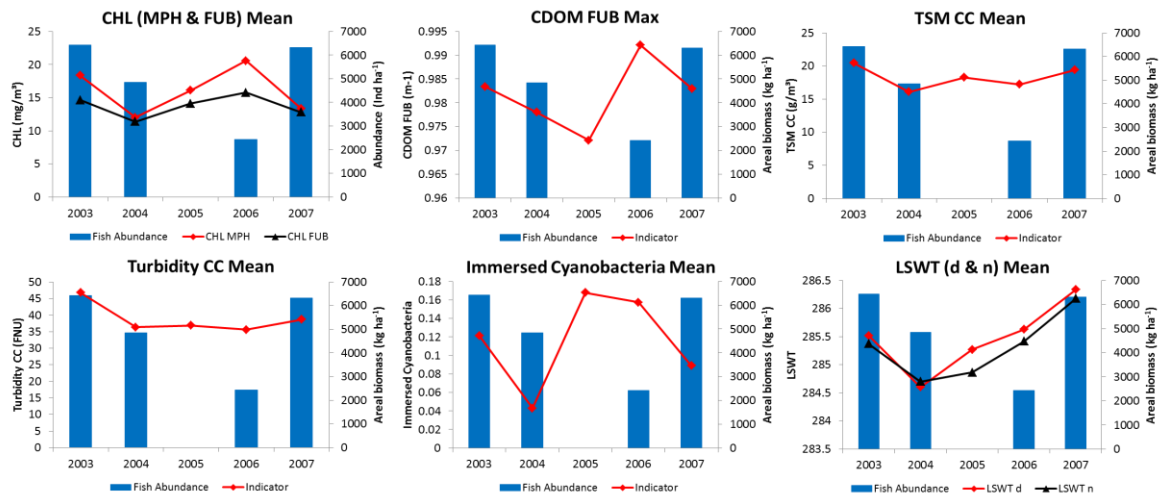


Figure 28 Yearly fish abundance and RS indicators for Lake Balaton trend analysis (Tátrai et al. 2008). Charts depict biodiversity measure (right Y axis, blue bars) and RS indicator (left Y axis, red and black lines), from 2003 to 2007 (X axis, no biodiversity data was available for 2005).

Cases of best match between biodiversity data from Tátrai et al. (2008) and RS indicators are displayed in Figure 28 and Figure 29. While visually some agreements can be discern (for example LSWT and fish abundance, or CHL and fish biomass), the fact that fish data for the year 2005 was unavailable does not allow us to correctly infer if a relationship between the two types of data does indeed occur.

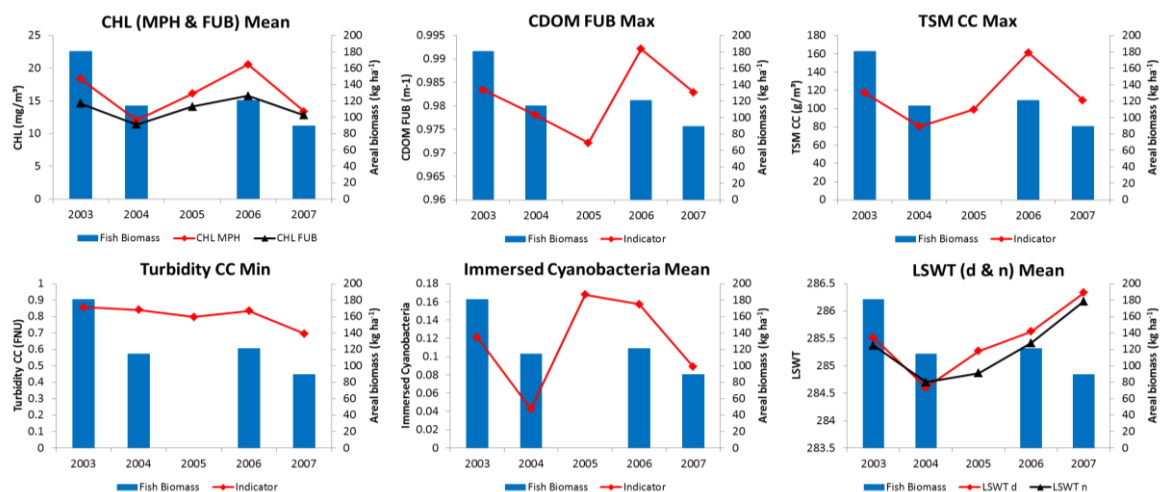


Figure 29: Yearly fish biomass and RS indicators for Lake Balaton trend analysis (Tátrai et al. 2008). Charts depict biodiversity measure (right Y axis, blue bars) and RS indicator (left Y axis, red and black lines), from 2003 to 2007 (X axis, no biodiversity data was available for 2005).

As for data from Balogh et al. (2008), a clear overlap between biodiversity values and RS indicators was not noticeable, nor were coherent patterns within the selected time series evident. Examples are given for the zebra mussel (Figure 31) and water birds data (Figure 30).

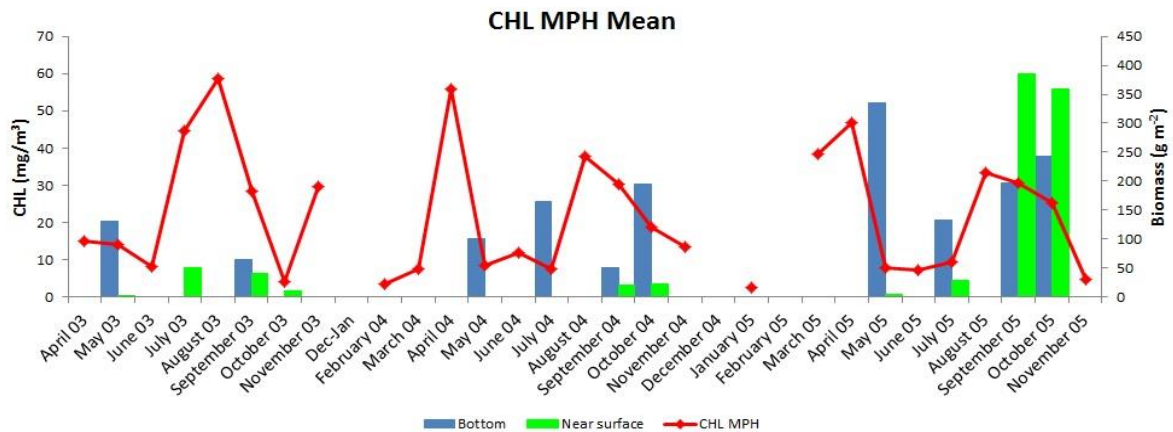


Figure 31: Monthly zebra mussel biomass and RS indicator for Lake Balaton trend analysis (Balogh et al. 2008). Charts depict biodiversity measure (right Y axis) - blue bars correspond to zebra mussel biomass collected at the bottom of the lake, while green bars correspond to biomass measurements conducted near the surface - and RS indicator (left Y axis, red line), from April 2003 to November 2005 (X axis).

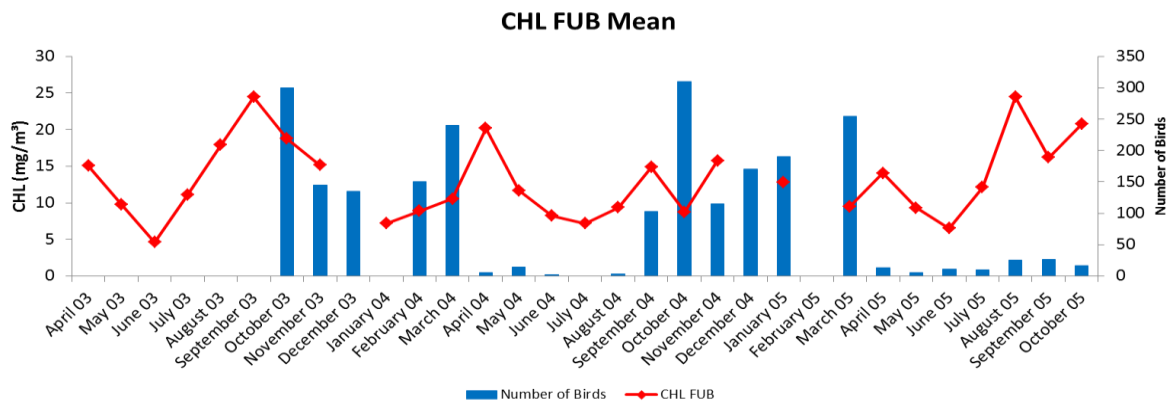


Figure 30: Monthly number of water birds and RS indicator for Lake Balaton trend analysis (Balogh et al. 2008). Charts depict biodiversity measure (right Y axis, blue bars) and RS indicator (left Y axis, red line), from April 2003 to October 2005 (X axis).

2.2.2.5 Lake Peipus

Trend data for Lake Peipus was extracted from Bobyrev et al. (2012) and consists of yearly data on fish catch of eight species, from 2002 to 2006. Cases of best match between biodiversity data from Bobyrev et al. (2012) and RS indicators are displayed in Figure 32 and Figure 33. As with the case of Lake Balaton, visually agreements can be discerned (both positive and negative relationship between the two types of data), but no statistically strong analyses were conducted due to the low sample size.

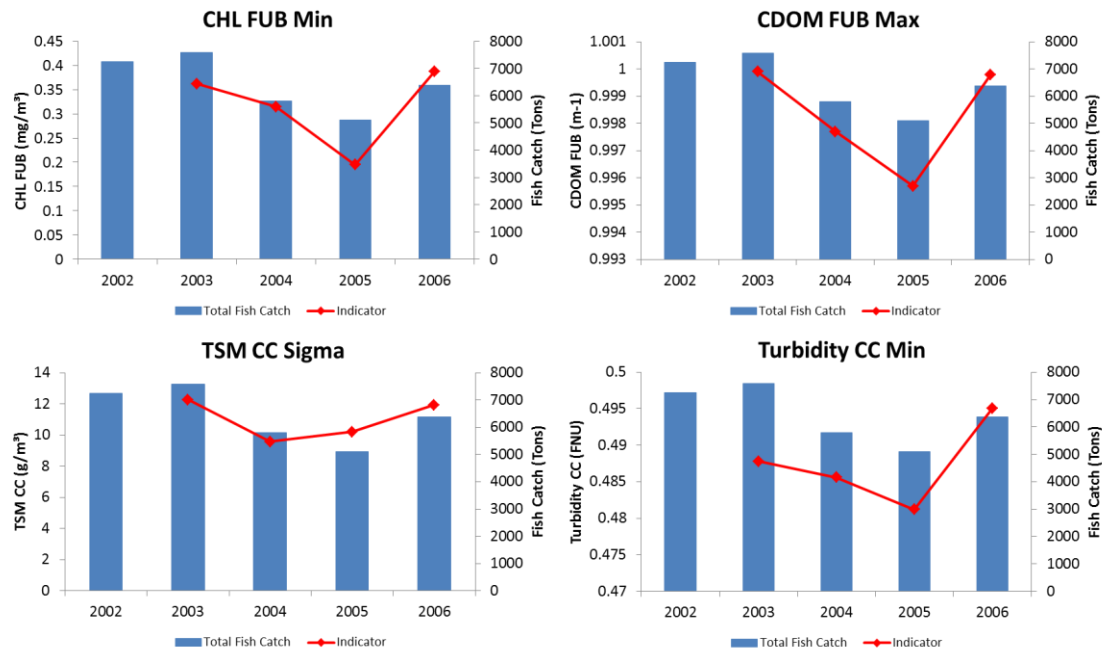


Figure 32: Yearly fish catch and RS indicators for Lake Peipus trend analysis (Bobyrev et al. 2012). Charts depict biodiversity measure (right Y axis, blue bars) and RS indicator (left Y axis, red line), from 2002 to 2006 (X axis).

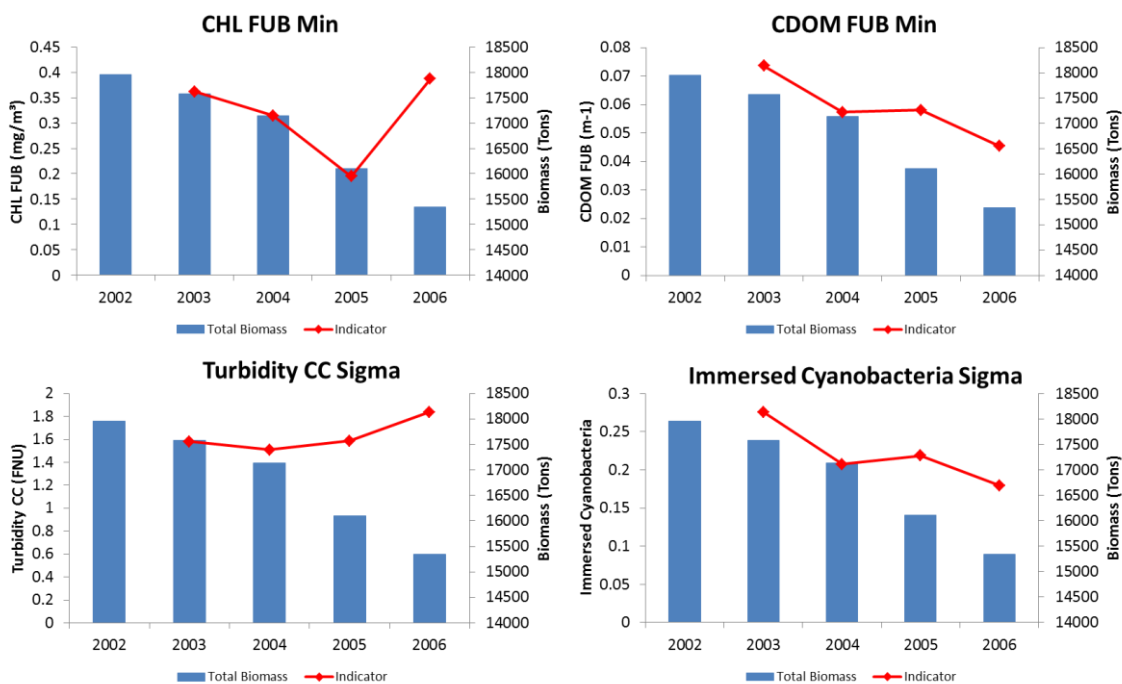


Figure 33: Yearly fish biomass and RS indicators for Lake Peipus trend analysis (Bobyrev et al. 2012). Charts depict biodiversity measure (right Y axis, blue bars) and RS indicator (left Y axis, red line), from 2002 to 2006 (X axis).

2.2.2.6 Lake Nicaragua

Trend data for Lake Nicaragua was extracted from Portocarrero (2013) and comprises yearly data on fish catch for six fish taxa, from 2002 to 2010. There were ten available RS indicators: CHL FUB, CHL MPH, CDOM FUB, TSM CC, Turbidity CC, immersed cyanobacteria, floating cyanobacteria and vegetation, and LSWT (d and n). In order to assess if there was any kind of agreement between biodiversity and indicator measurements we conducted Spearman's rank order correlation analyses between RS indicators – including the mean, minimum, maximum and standard deviation (sigma) - and fish catch data. Significance of results were tested by their probability values (p), considering a confidence level of $p < 0.05$.

Six statistically relevant correlations were obtained between biodiversity data and RS indicators (Table 7). Positive correlation were found for CHL MPH sigma, CHL FUB minimum and turbidity CC sigma, while for CHL FUB maximum, CDOM FUB mean and LSWT n sigma the relationship was negative (Figure 34 and Figure 35).

Table 7: Correlation analysis for Lake Nicaragua trend data. Correlation values are based on Spearman's r . Values above zero indicate positive correlations, while below zero indicate negative correlations. Probabilities are depicted between brackets for the highest correlation values (≥ 0.5 or ≤ -0.5), which are also displayed in Figure 34.

Indicators	Fish Catch
CHL MPH	Min -0.37
	Max -0.24
	Mean -0.45
	Sigma 0.67 (0.071)
CHL FUB	Min 0.67 (0.071)
	Max -0.5 (0.207)
	Mean 0.1
	Sigma -0.29
CDOM FUB	Min 0.38
	Max -0.17
	Mean -0.67 (0.071)
	Sigma 0.21
TSM CC	Min -0.33
	Max -0.4
	Mean -0.26
	Sigma 0.14
Turbidity CC	Min -0.1
	Max -0.02
	Mean 0.36
	Sigma 0.52 (0.183)
Immersed Cyanobacteria	Mean -0.4
	Sigma 0.48
Floating Cyanobacteria	Sigma -0.38
Floating Vegetation	Max -0.42

Indicators		Fish Catch
LSWT d	Sigma	-0.32
	Min	0.07
	Max	0.26
	Mean	0.14
	Sigma	0.21
LSWT n	Min	0.19
	Max	-0.07
	Mean	0.1
	Sigma	-0.57 (0.139)

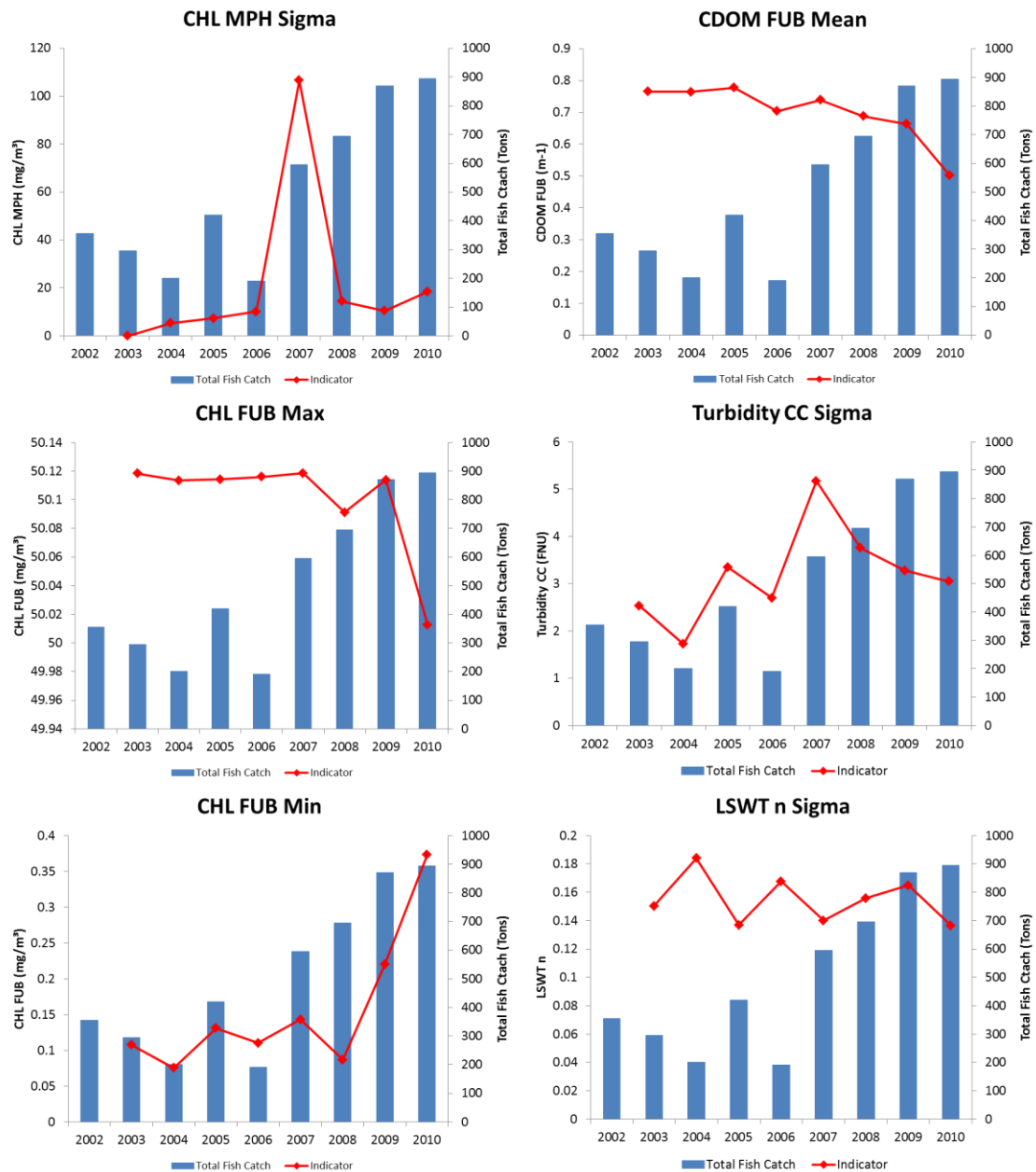


Figure 34: Yearly fish catch and RS indicators for Lake Nicaragua trend analysis (Portocarrero 2013). Charts depict biodiversity measure (right Y axis, blue bars) and RS indicator (left Y axis, red line), from 2002 to 2010 (X axis).

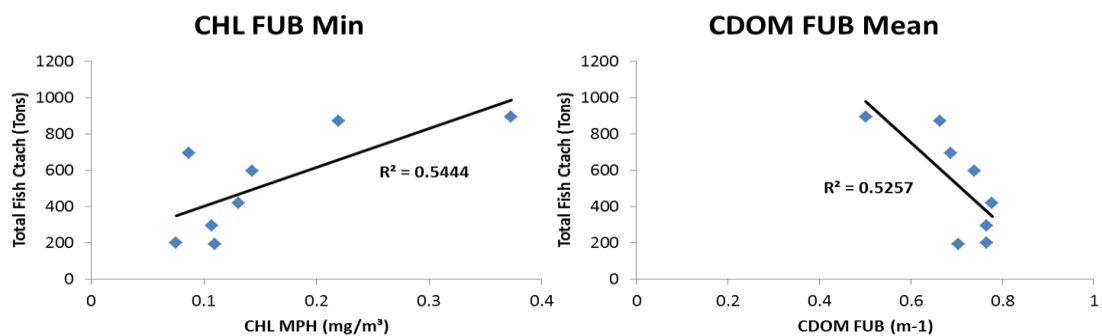


Figure 35: Statistically relevant correlations between indicators and biodiversity data based on Spearman's r. Scatter plots depict fish capture data (Y axis) and RS indicators (X axis), per year (blue diamond shapes).

2.2.2.7 Lake Tahoe

Trend data for Lake Tahoe was retrieved from Wittmann et al. (2012), and comprises data on the abundance and diversity of macro-invertebrates for the months of March, April, July, September and December of 2009, and April and July of 2010.

The paucity of biodiversity data did not allowed us to conduct any type of analyses and no particular pattern is discernible from the graphics (Figure 36, as an example).

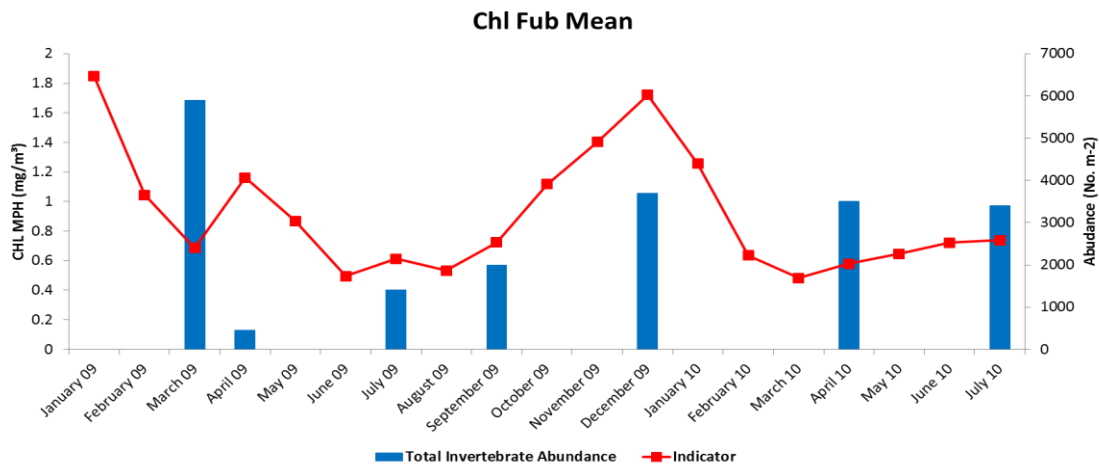


Figure 36: Monthly invertebrate biomass and RS indicator for Lake Tahoe trend analysis (Wittmann et al. 2012). Chart depicts biodiversity measure (right Y axis, blue bars) and RS indicator (left Y axis, red line), from January 2009 to July 2010 (X axis).

2.2.2.8 Lake Päijänne

Trend data for Lake Päijänne was retrieved from Syrjänen & Valkeajärvi (2010), and comprises data on yearly density of age-0 brown trout in two tributaries of Lake Päijänne, Muuramenjoki and Rutajoki, from 2002 to 2008.

The ten RS indicators were available and data was extracted for the two tributaries, however we only used data from five (CHL FUB, CHL MPH, CDOM FUB, TSM CC and Turbidity CC), since remaining indicators showed no variability (maybe related with the small area of the tributaries). Again, in order to assess if there was any kind of agreement between biodiversity and indicator measurements we conducted Spearman's rank order correlation analyses between RS indicators – including the mean, minimum, maximum and standard deviation (sigma) - and fish density data. Significance of results were tested by their probability values (p), considering a confidence level of $p < 0.05$.

Table 8: Correlation analysis for Lake Päijänne trend data. Correlation values are based on Spearman's r . Values above zero indicate positive correlations, while below zero indicate negative correlations. Probabilities are depicted between brackets for the highest correlation values (≥ 0.5 or ≤ -0.5).

Indicators		Muuramenjoki	Rutajoki
CHL MPH	Min	-0.52 (0.288)	-0.66 (0.156)
	Max	-0.75 (0.084)	-0.26
	Mean	-0.52 (0.288)	-0.09
	Sigma	-0.67 (0.148)	-0.09
CHL FUB	Min	-0.67 (0.148)	-0.83 (0.042)
	Max	-0.32	-0.2
	Mean	-0.84 (0.036)	-0.31
	Sigma	-0.61 (0.2)	-0.2
CDOM FUB	Min	-0.75 (0.084)	-0.37
	Max	0.67 (0.148)	0.37
	Mean	-0.75 (0.084)	-0.6
	Sigma	0.32	-0.26
TSM CC	Min	-0.58 (0.288)	-0.83 (0.042)
	Max	0.78 (0.066)	-0.26
	Mean	-0.12	-0.31
	Sigma	0.46	-0.49
Turbidity CC	Min	-0.38	-0.2
	Max	0.17	-0.31
	Mean	-0.12	-0.03
	Sigma	-0.23	-0.31

For the Muuramenjoki tributary, 12 statistically relevant correlations were retrieved, while for Rutamenjoki only three (Table 8). Figure 37 and Figure 38 illustrate the best examples. The great majority of indicators are negatively correlated with fish densities, exceptions made for CDOM FUB Max and TSM CC Max. For Muuramenjoki, CHL FUB Mean and

CDOM FUB Min show a strong negative relationship, while the same applies for CHL FUB Min in the Rutamenjoki tributary.

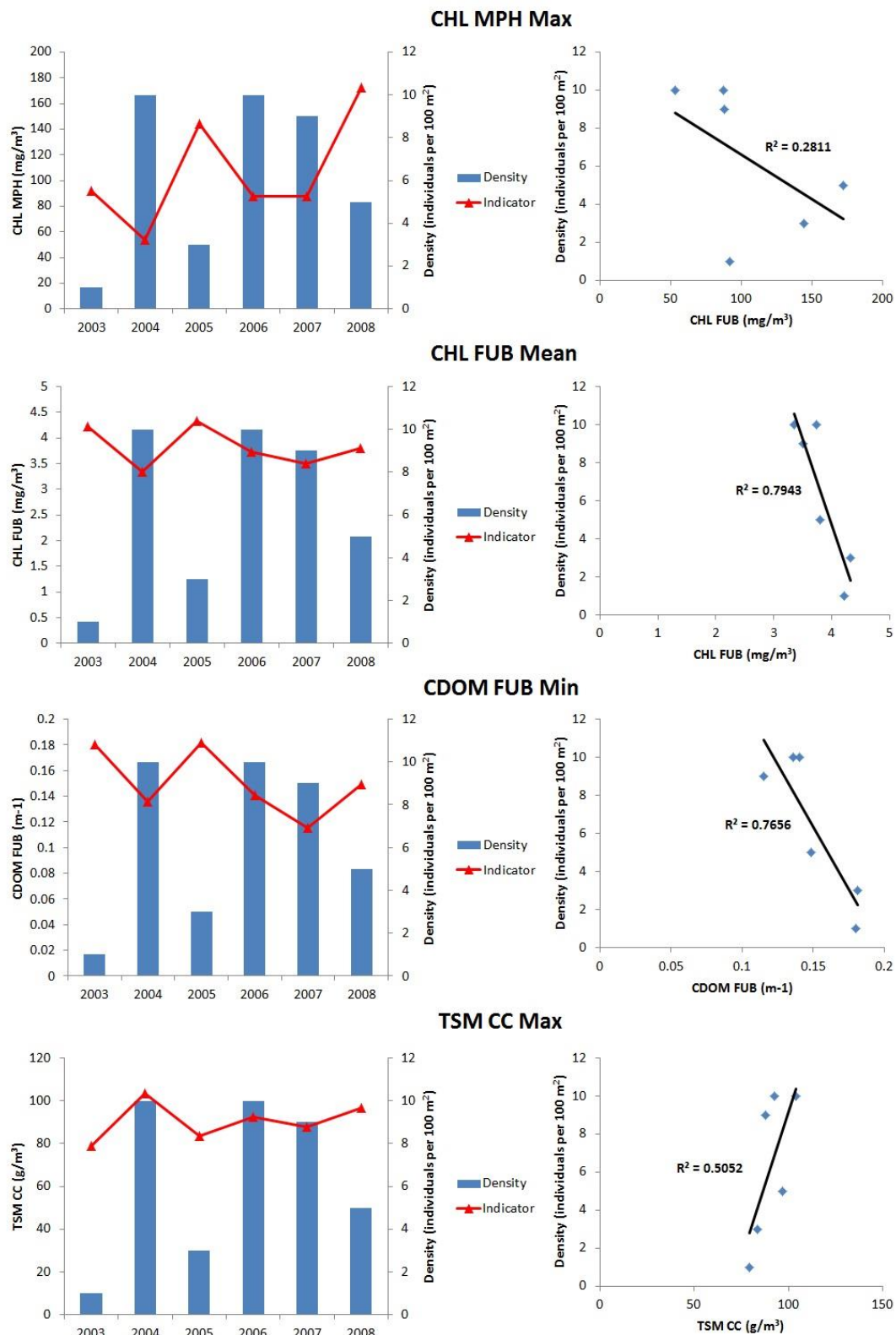


Figure 37: Yearly fish catch and RS indicators for the Muuramenjoki tributary, Lake Päijänne trend analysis (Syrjänen & Valkeajärvi 2010). On the left side, charts depict biodiversity measure (right Y axis, blue bars) and RS indicator (left Y axis, red line), from 2003 to 2008 (X axis). On the right side, scatter plots depict fish capture data (Y axis) and RS indicators (X axis), per year (blue diamond shapes).

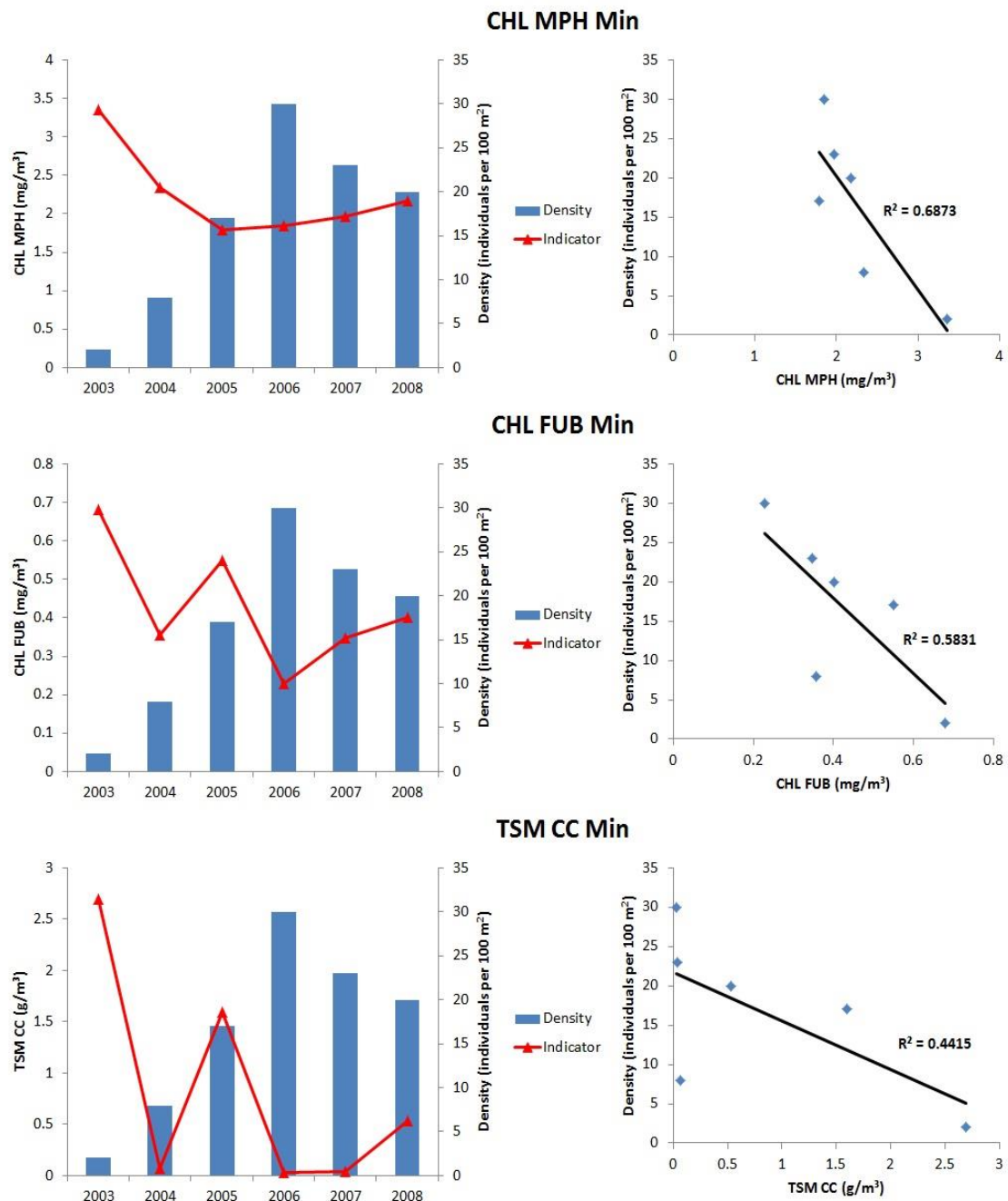


Figure 38: Yearly fish catch and RS indicators for the Rutajoki tributary, Lake Päijänne trend analysis (Syrjänen & Valkeajärvi 2010). On the left side, charts depict biodiversity measure (right Y axis, blue bars) and RS indicator (left Y axis, red line), from 2003 to 2008 (X axis). On the right side, scatter plots depict fish capture data (Y axis) and RS indicators (X axis), per year (blue diamond shapes).

2.2.2.9 Lake Victoria

Trend data for Lake Victoria was retrieved from Kayanda et al. (2009), and comprises data on the abundance and diversity of macro-invertebrates for the months of March, April, July, September and December of 2009, and April and July of 2010.

As with Lake Tahoe, the paucity of biodiversity data did not allowed us to conduct any type of analyses and no particular pattern is discernible from the graphics (Figure 39, as an example).

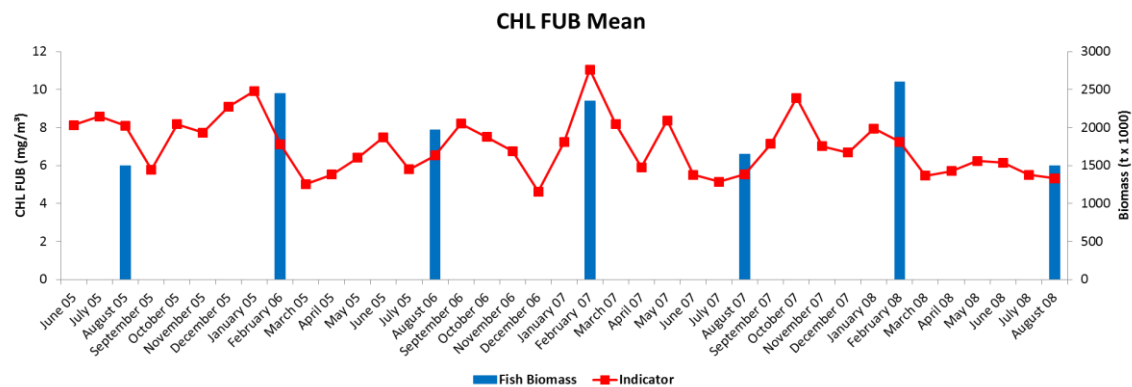


Figure 39: Monthly fish biomass and RS indicator for Lake Victoria trend analysis (Kayanda et al. 2009). Chart depicts biodiversity measure (right Y axis, blue bars) and RS indicator (left Y axis, red line), from June 2005 to August 2010 (X axis).

3 Drylands

To assess the quality of the full set of DIVERSITY II products, a series of test sites have been selected. These sites will be used to validate project products with respect to the consolidated user requirements. An extensive evaluation of the product quality and representativeness will be performed, through a validation on a representative set of selected five dryland sites (Figure 40).

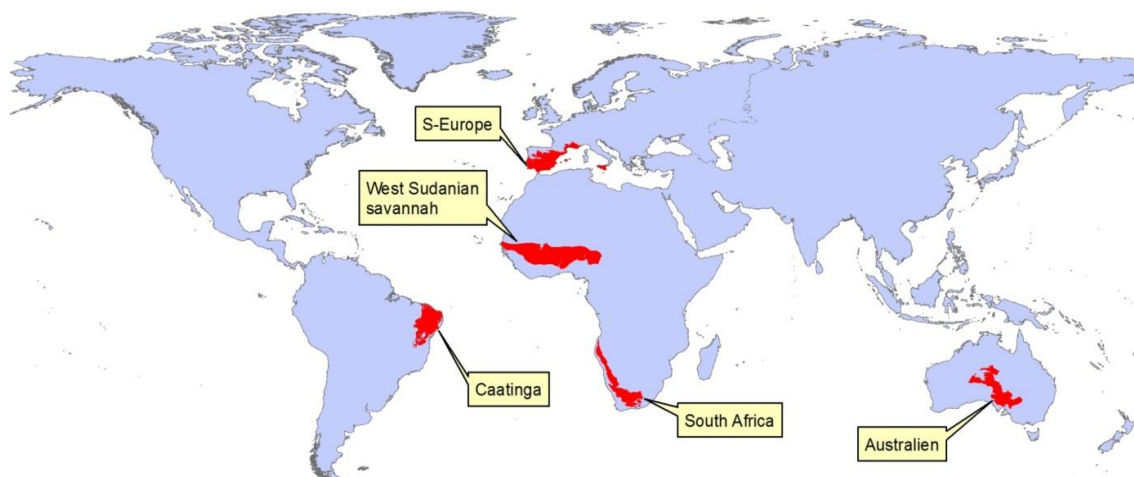


Figure 40: Global map depicting the location of the five dryland test sites.

Selection criteria for test sites were made according to the SoW, which stresses the need to concentrate efforts on biodiversity hotspots and other areas of specific value for the conservation of biodiversity. Additionally, dryland test sites should include large and representative areas of dryland-related biomes from the terrestrial ecoregions defined by the WWF (<http://worldwildlife.org/biome-categories/terrestrial-ecoregions>) and have available reference projects.

Descriptive parameters for each of the five dryland test-sites are displayed in Table 9, including: the terrestrial ecoregion to which the test-site belongs (WWF 2013a,b,c,d,e,f,g,h,i); if the test site is part of a biodiversity hotspot; the area and countries covered by the test; and the central latitude and longitude of the test site.

Table 9: Descriptive parameters of the five dryland test-sites.

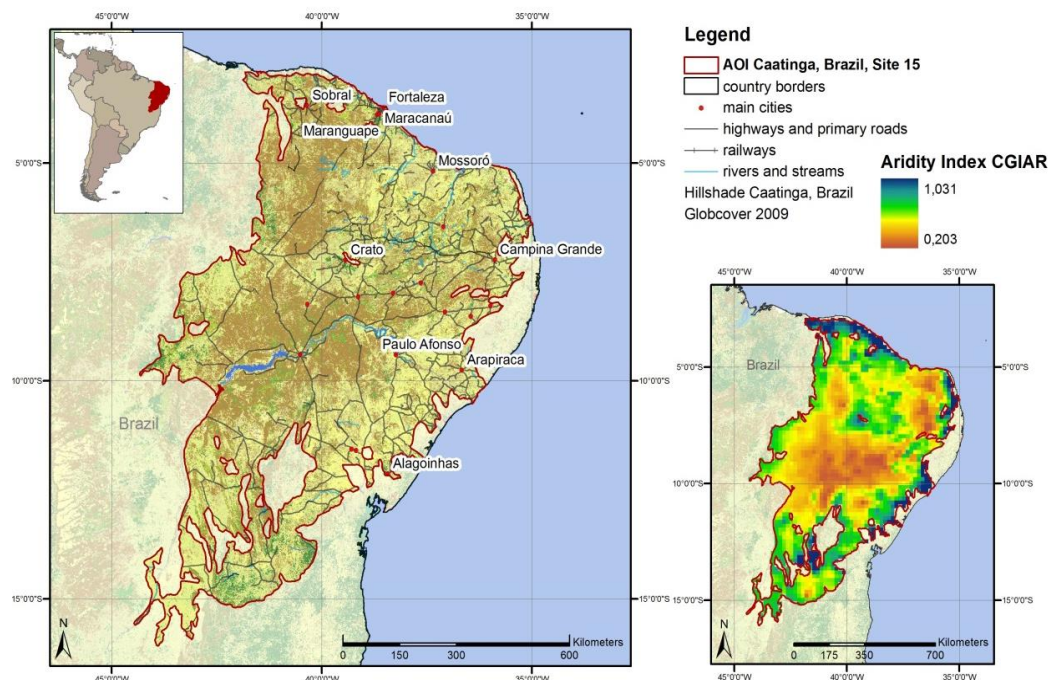
Dryland	Eco-region	Terrestrial ecoregion	Biodiversity Hotspot	Area (km ²)	Country	Lat.	Lon.
S-Europe	10	Iberian conifer forests, North-eastern Spain and Southern France Mediterranean forests, Iberian sclerophyllous and semi-deciduous forests, Southwest Iberian Mediterranean sclerophyllous and mixed forests, Northwest Iberian montane forests	Yes	524,698	Portugal, Spain, France, Italy	40.00	-3.00
South	12	Namibian savannah	Partially	576,085	Angola,	-26.00	17.55

Dryland	Eco-region	Terrestrial ecoregion	Biodiversity Hotspot	Area (km ²)	Country	Lat.	Lon.
Africa		woodlands, Nama Karoo			Namibia, South Africa		
West Sudanian Savannah	13	West Sudanian Savannah	-	1,641,911	Senegal to Nigeria	12.00	-2.00
Caatinga	15	Caatinga	-	718,135	Brazil	-39.50	-8.50
Australien	20	Tirari-Sturt stony desert, Central Ranges xeric scrub	-	660,019	Australia	-28.70	135.50

3.1 Test Sites

In the following sections, descriptive parameters for each test site and broad characterisations of the sites and surrounding areas are presented. Focus is given to the geographic location, climatic conditions, hydrological and topographical features, geologic background, biodiversity features (fauna and flora), important threats to the environment, present levels of protection (Protected Areas), and conservation/management practices in the various regions.

3.1.1 Caatinga



Cartographic References: Projection: GCS_WGS_1984 / Datum: WGS 1984

Figure 41: Dryland test site – Caatinga.

The Caatinga is the only endemic biome of Brazil and one of the world's 37 major wilderness areas, comprising a mosaic of scrub vegetation and dry forest (Leal et al. 2005). Located in the semi-arid Northeast (Figure 41), it occupies an area of around 800,000 km² that represents over 10% of the Brazilian territory, extending across eight federal states: Piauí, Ceará, Rio Grande do Norte, Paraíba, Pernambuco, Alagoas, Sergipe, Bahia, and northern Minas Gerais (MMA 2007; Camardelli et al. 2012). The Caatinga is mainly settled on a flattened depression of sandstone and crystalline rock, broken by isolated plateaus, and mountain massifs that can reach elevations of over 2000 m above sea level (Leal et al. 2005; Santos et al. 2011; WWF 2013a). The region is submitted to a hot and dry climate, with averages of annual temperature between 24-26°C, annual rainfall ranging from 240 to 1500 mm and high evapotranspiration rates of over 1400 mm (Sampaio 1995; Prado 2003; Auler et al. 2004; WWF 2013a). The climate is seasonal with a well-defined rainy season during the Summer/Autumn, followed by 6 to 11 dry months (larger number in the more central areas) (Auler et al. 2004; Leal et al. 2005). Additionally, interannual variability in rainfall is considerably high, resulting in periodic severe droughts (Leal et al. 2005).

For long the Caatinga has been perceived as holding very few species, but despite the harsh climate, the region harbours a great degree of biological diversity. Substantial biotic interchange has always occurred between the Caatinga and other major Neotropical biomes that border the region, such as the Amazon forest to the northwest, and the Cerrado and Atlantic forest to the west and southwest (WWF 2013a). Adding to this is the Caatinga's own heterogeneous landscape, which has allowed for example the existence of "moist forest islands" of both Amazon and/or Atlantic forest affinity, with great levels of endemism, in higher elevation topographic features such as plateaus and mountain ranges (Camardelli et al. 2012).

Besides the relict moist forests, the Caatinga is predominantly characterized by steppe savannah and dry forest type habitats, with intermittent river systems (Auler et al. 2004; Leal et al. 2005). The vegetation is made up mainly of xerophytic, woody, thorny, and deciduous physiognomies, which lie dormant and leafless throughout most of the year (Auler et al. 2004; Leal et al. 2005). Inventories and assessments have recorded up to 1200 species of vascular plants, of which around 30% are only found in the region (Leal et al. 2005; WWF 2013a).

In terms of animal taxa, the numbers yielded by the Caatinga are also considerable, with at least 240 fish species, 175 amphibians and reptiles, 520 birds (across 62 families) and 156 mammals (of which 44.7% are bats) (Oliveira et al. 2003; Santos et al. 2011; Albuquerque et al. 2012; Sá-Neto & Marinho-Filho 2013). Invertebrates are still poorly studied, however there are no reasons to believe that their numbers and diversity will be low (Araújo et al. 2005), and for example 187 bee species have been reported in the region (Santos et al. 2011). Overall levels of endemism are not as high as for plants, but still, values range from about 7% for mammals, 3 to 9 % for birds, 12% for amphibians and reptiles, and 57% for fishes (Leal et al. 2005; Santos et al. 2011; Albuquerque et al. 2012). All of these current biodiversity scores are equal or higher when compared to other semi-arid biomes around the globe, and numbers can increase even further if we take into account that by 2004/2005 41% of the region had never been surveyed and 80% had been only poorly explored (Leal et al. 2005; Santos et al. 2011). Overall, sample size and sampling efforts are relatively small and scattered, which has made many researchers adhere to information networks such as TROPIC-DRY to better maintain

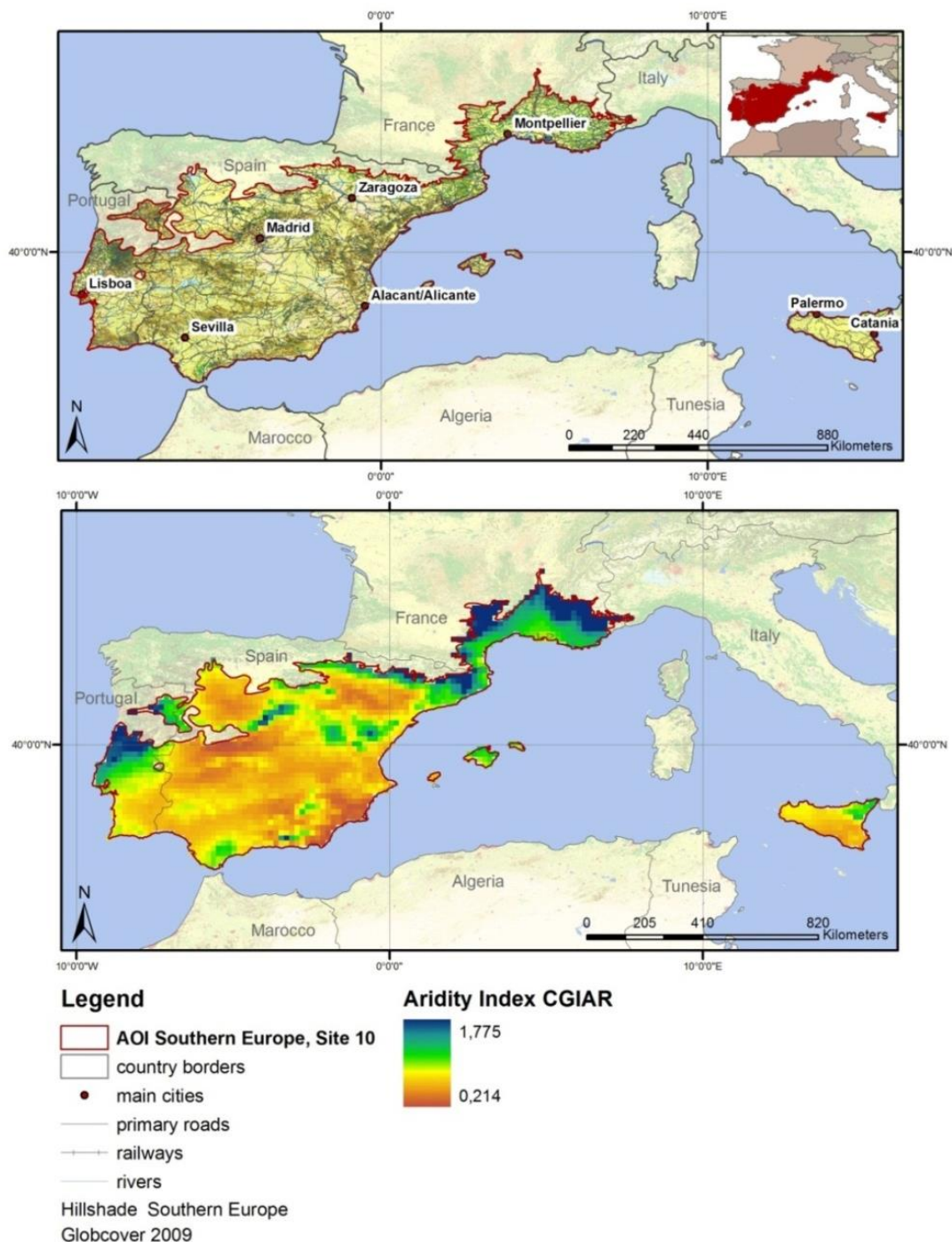
contact with other works and to compare results to other dry forest regions worldwide (Albuquerque et al. 2012).

Endemic species of the Caatinga are highly threatened. The more serious cases come from bird species: the Spix's macaw (*Cyanopsitta spixii*) has become extinct in the wild and it is maintained through captive breeding programs; the Lear's macaw (*Anodorhynchus leari*) is reduced to a population of around 1000 individuals, separated into only two colonies (Leal et al. 2005). Another example comes from the Brazilian three-banded armadillo (*Tolypeutes tricinctus*), which was also thought to be extinct but was recently rediscovered in remnants of relict forest (Leal et al. 2005).

The major threat for the conservation of the Caatinga and its wildlife are, without a doubt, the destruction and transformation of habitat for intensive land uses. Over 25 million people live in the Caatinga, most of them poor, and around 50% of the region has been already either completely converted, or altered in a significant way, due to the establishment of unsustainable agriculture and livestock practices (including exotic species), extraction of wood for fuel, uncontrolled fires, modification of river regimes and water flows, and construction of cities and roads (Leal et al. 2005; Alves et al. 2009; WWF 2013a). The intensive land use is responsible for severe environmental damage, with 15% of the region now under the risk of desertification, and remaining natural habitats being highly fragmented and degraded (Santos & Tabarelli 2002; Leal et al. 2005). Another major concern for the sustainability of the Caatinga's biota is intense hunting and persecution. This includes both: illegal hunting and sale of endangered wildlife; and subsistence hunting by the local people, for which various animals are important resources of food but also of medicinal remedies and ornamental pieces, as well as some species being killed for being considered threats (mainly snakes), while others are kept and enjoyed as pets (birds mostly) (Alves et al. 2009; Barbosa et al. 2011; Albuquerque et al. 2012).

In the face of all these sizable threats and increasing human pressures, the Caatinga has the fewest and smallest protected areas of any major Brazilian biome, with only 11 reserves that cover less than one per cent of the region (Leal et al. 2005). More recently in 2007, the Brazilian Ministry of the Environment (MMA) defined priority conservation areas across the entire country, and a study in 2012 showed that the majority of areas of endemism for amphibians overlapped with the priority areas set by the MMA in the Caatinga biome (Camardelli et al. 2012). Despite the current state of knowledge and conservation practice in the Caatinga, there is a clear effort from various government, state and local agencies, as well as NGO's, to try turn the tide, and a good example comes from the Parque Nacional da Serra da Capivara, one of the most visited and best managed National Parks that insures the conservation of a key set of species as well as an important archaeological site (Leal et al. 2005; Santos et al. 2011; Albuquerque et al. 2012; WWF 2013a). Additionally, four AZEs sites are located within the Caatinga (AZE 2013), and the north-eastern part is considered an Important Bird Area (BirdLife International 2013i).

3.1.2 Southern Europe



Cartographic References: Projection: GCS_WGS_1984 / Datum: WGS 1984

Figure 42: Dryland test site – Southern Europe.

The study area considered here covers the north-western part of the Mediterranean basin, and includes the Iberian Peninsula (except for the more Atlantic influenced northwest), the Balearic Islands, southern France and the island of Sicily (Figure 42). The area encompasses many types of ecoregions (WWF 2013b,c,d,e,f), but overall it is characterized by either oak tree, coniferous and deciduous forest where water availability is relatively higher, or sclerophyllous shrublands in dryer regions (Derneği 2010). Altogether the area is subjected to

the typical Mediterranean type climate, with cool and wet winters followed by hot and dry summers (Blondel et al. 2010). Annual average temperatures may vary between 8-19°C, and annual precipitation from 300-900 mm (WWF 2013b,c,d,e,f), although the Mediterranean basin is known for holding very distinct local microclimates, and the climatic conditions can be very unpredictable within one year and/or during successive years (Blondel et al. 2010). Torrential rainfalls are quite common during autumn/winter that can lead to violent floods, while in the summer the availability of surface water is very low and severe droughts can last for up to two months (Blondel et al. 2010).

The topography of the region certainly contributes for the climatic variability. The region contains sand dunes, cliffs and salt lagoon systems along the coastal strip, wide plains and plateaus, deep river valleys and extensive mountain chains, not to mention the many island systems (Blondel et al. 2010; WWF 2013b,c,d,e,f). In terms of the geological background, the region is predominantly composed of Mesozoic and Quaternary sedimentary rock with some extents such as the central plateau between Portugal and Spain (the Iberian massif) and areas in the Balearic Islands constituted by older crystalline rock (WWF 2013b,c,e,f). Also important to mention is the active Etna Volcano in Sicily, and consequently the volcanic rock that characterizes that area (WWF 2013d). The Mediterranean basin has a complex geological history, and the region has gone through many great scale tectonic and orogenic events, one of the most recent being the Messinian Salinity Crisis around 5.5 million years ago (Govers 2009; Blondel et al. 2010).

With such a complex geographic setting, the Mediterranean basin harbours an astonishing biodiversity. It was one of the first regions to be recognized into the 25 Global Biodiversity Hotspots (Myers et al. 2000), of which it is the third richest in terms of plant diversity (Mittermeier et al. 2004). The north-western Mediterranean vegetation is composed of a mix of evergreen, deciduous and conifer tree species, like wild olive (*Olea europaea*), oak trees (*Quercus* spp.) and pine trees (*Pinus* spp.) (WWF 2013c). Shrublands, or “maquis”, include species such as *Myrtus communis*, *Juniperus phoenicea* and *Chamaerops humilis*, while in river beds we can find *Tamarix* spp., *Fraxinus angustifolia*, *Salix* spp., among many others (WWF 2013c,f). The region includes many centres of plant diversity and endemism rates go from 10 to over 20 %. The Balearic Islands host 180 endemic plant species (out of 1450), while Sicily is home for 310 endemics (out of 2700 species) (WWF 2013d,f).

Faunal richness is also high in north-western Mediterranean. While mammals and birds are mostly of Eurasian and African origin, the amphibian, reptile and freshwater fish fauna present considerable rates of endemism (Derneği 2010; Cox et al. 2006). Rivers in the region harbour unique fish species like *Aphanius iberus*, *Anaocypris hispanica* and *Barbus microcephalus* (WWF 2013b,c). In terms of amphibians, we have the examples of the Mediterranean tree frog (*Hyla meridionalis*), the Western spadefoot toad (*Pelobates cultripes*) and the Iberian ribbed newt (*Pleurodeles waltl*) (Gasc et al. 1997; Cox et al. 2006), and for reptiles we have examples such as of the Iberian worm lizard (*Blanus cinereus* - only member of the amphisbaenians found in Europe), the Mediterranean turtle (*Mauremys leprosa*), many species of lizards (*Iberolacerta* spp., *Podarcis* spp., *Psammodromus* spp.) and of snakes/vipers (*Natrix* spp., *Vipera* spp.), among many other examples of herpetofauna (Gasc et al. 1997; Cox et al. 2006). The region is also home for emblematic endemic species of mammals such as the Iberian lynx (*Lynx pardinus*) and the rare Sicilian shrew (*Crocidura sicula*), as well as important populations of otter (*Lutra lutra*) and European polecat (*Mustela putorius*) (WWF 2013b,c,d,f).

Finally, bird diversity is also tremendous, and the region constitutes an important wintering and nesting ground for countless species. Important populations of endangered raptor species can be found in the area, like the black vulture (*Aegypius monachus*), griffon vulture (*Gyps fulvus*) and imperial eagle (*Aquila heliaca*) (WWF 2013c,f). Another endangered species present in the region is the great bustard (*Otis tarda*) (WWF 2013b). Some delta areas can harbour up to 30,000 pairs of water birds that include many ducks (*Anas* spp.), gulls (*Larus* spp.), terns (*Sterna* sp.) and countless other species (WWF 2013c). The region even holds the only European populations of flamingos (*Phoenicopterus ruber*) (WWF 2013c,e).

The Mediterranean basin is one of the richest regions in terms of biodiversity, but also one of the most threatened. For several millennia, the region has sustained some of the greatest civilization in History (Derneži 2010). The landscape has been severely modified and nowadays very little pristine habitat spots still exist (Blondel et al. 2010; Derneži 2010). Then again, this has given time for the fauna and flora of the region to adapt to the new altered environmental settings, and some species are now dependent on the coexistence with humans, mainly in agricultural systems (Cox et al. 2006; Blondel et al. 2010). The greatest threat to biodiversity has been, without a doubt, the continuous destruction, fragmentation and modification of habitat, whether it is for agricultural use, urban or industrial development. For example, the intensification of agricultural practices has been responsible for the loss of wetlands that are essential for the survival of amphibian species (Ferreira & Beja, 2013). The entire Mediterranean basin is home for more than 400 million people, and future predictions show an additional increase. Furthermore, the Mediterranean countries are international tourism destinations, receiving around 200 million people per year, mainly along the coast line (Cox et al. 2006; Cuttelold et al. 2008). So establishing conservation efforts is always an arduous endeavour due to the variety of conflicts it can generate (Derneži 2010). Another major issue is water availability. The combination of dry summers, unsustainable farming practices and construction of river dams, result in dried up river beds that affect freshwater habitat species, and desertification, degradation and erosion of the land (60% of Portugal for example faces a moderate risk of desertification) (Cox et al. 2006; Cuttelold et al. 2008). All of the above pressures will become even more problematic as a result of global climate change, and forest fires and severe droughts will increase in frequency and severity (Cuttelold et al. 2008).

The Mediterranean basin is considered as one of the Global Biodiversity Hotspots (CI 2013b), and fortunately in the last decades, efforts made by various institutions, from the European Union to more local forms of government, not to mention NGOs, have been responsible for the emergence of numerous conservation sites that include the Natura 2000 sites, national and regional wildlife parks and reserves (Cuttelold et al. 2008). A good example is the Doñana National Park in Spain, which holds the remaining wild population of the Iberian lynx. Additionally the test site holds one AZE site (AZE 2013) and many Ramsar sites (Ramsar 2013) and IBAs (BirdLife International 2013). Conservation sites alone do not ensure the protection of biodiversity, but together with species specific actions, continuous research and monitoring, as well as better education and communication with the general public, many species have been saved from extinction (Cuttelold et al. 2008).

3.1.3 West Sudanian Savannah

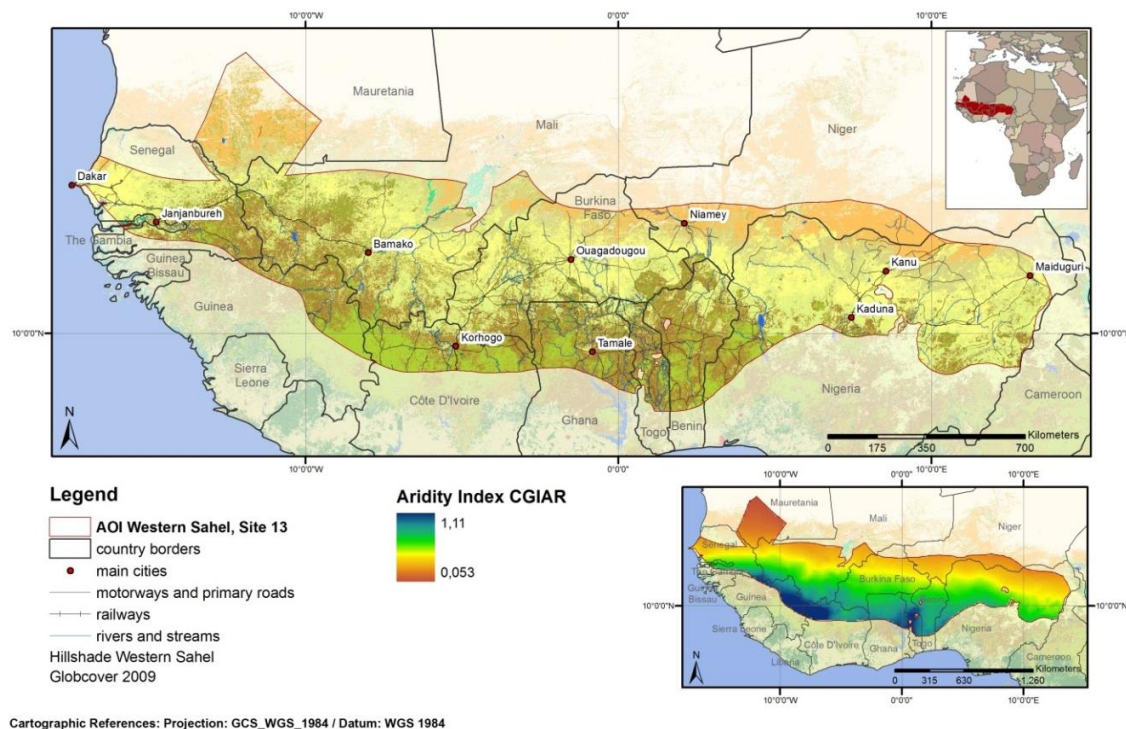


Figure 43: Dryland test site – West Sudanian Savannah.

The West Sudanian Savannah is located in West Africa, constituting a stretch of hot, dry and wooded savannah that is delimited by the Atlantic Ocean to the West, the Sahel to the North, the Guinean forest-savannah mosaic to the South and the eastern border of Nigeria to the East (Figure 43) (Chidumayo et al. 2011; WWF 2013g). It covers parts of the states of Senegal, Gambia, Mauritania, Mali, Guinea, Côte d'Ivoire, Burkina-Faso, Ghana, Togo, Benin, Niger and Nigeria. The region is subjected to a tropical and strongly seasonal climate, where mean monthly maximum and minimum temperatures are around 32°C and 20°C, respectively (Bie 1991; WWF 2013g). Annual rainfalls are limited to one season that starts by May-June and ends September-October, with values of precipitation varying between 600 mm in the northern parts of the savannah and 1200 mm in the south (Bie 1991). The most important limiting factor to ecosystem functioning is the length and severity of the dry season that can last two to eight months, with temperatures being high year-round and rainfalls being more variable and irregular, especially in the northern borders of the Sahelian region (Bie 1991). Climatic variations across different years are also common, which are responsible for very prolonged and harsh droughts, especially since the 1960-70's (Bie 1991; Chidumayo et al. 2011). In terms of landscape, the region is mainly flat with no significant topographic features (elevation usually does not surpass 700 m). Overlying the region are mainly Precambrian and Cambrian rocks, in the form of granitic shields and sandstone (Bie 1991).

The West Sudanian Savannah's vegetation is mainly characterised by woodland savannah species, a mix of woody trees and tall grasses. Trees rarely exceed 10 m in height, in the north of the savannah mainly belonging to the genus *Combretum* spp. and *Acacia* sp., and as we go south *Tamarindus indica*, *Prosopis africana* and *Commiphora africana* begin to dominate the landscape. Further south in wetter areas, we can find *Terminalia* spp., *Isobrerlinia doka* and *Azalia africana* (Bie 1991; WWF 2013g). The herbaceous layer is constituted

predominantly by *Hyparrhenia* spp. (elephant grass that can reach up to 4 m in height), but also by *Andropogon* spp., *Ctenium* spp. and *Loudetia* spp. (Bie 1991; WWF 2013g).

The savannah has also a considerably rich fauna, but endemism is low as the majority of species have pan-African distributions. Endemic species include two small mammals (*Crocodyra cinderella* and *Lemniscomys limulus*), eight reptiles and three amphibians (Ba et al. 2012; WWF 2013g). Mammals are especially well represented and the region includes populations of threatened large-sized species like the giant eland (*Taurotragus derbianus*), African wild dog (*Lycaon pictus*), cheetah (*Acinonyx jubatus*), lion (*Panthera leo*), elephant (*Loxodonta africana*), western giraffe (*Giraffa camelopardalis peralta*) and West African Savannah buffalo (*Syncerus caffer brachyceros*), most of them restricted to protected areas (Bouche et al. 2011; Riggio et al. 2013; WWF 2013g). Invertebrates must also be very diverse in the region but remain to quantify. Odonate (dragonflies and damselflies) diversity is lower in the savannah as compared to forest but still hold endemic species like *Zygonychidium gracilebeing* (Dijkstra & Vick 2004). Termites of the genus *Macrotermes* also provide relevant services to the ecosystem, as their abandoned mounds are colonized by many plant species and act as source of landscape heterogeneity (Kirchmair et al. 2012).

Threats to biodiversity in the West Sudanian Savannah are intimately related with poverty, which drive governments and local people to make short-term decisions that often hamper long-term conservation measures (Paré et al. 2009). The region has been exhibiting a high and rapid population and urbanization growth (3.1 and 7% per year, respectively), yet still, more than half of the population is dependent on agriculture and herding (Chidumayo et al. 2011). The demand and transformation of land for agricultural practices have reduced, degraded and fragmented natural habitat (WWF 2013g). One particular type of culture, cotton, has been expanding tremendously and since it requires large doses of pesticides, it has very harmful effects over the environment (Baudron et al. 2009). Grazing animals, cutting trees for fuelwood and charcoal, setting of uncontrolled fires (natural fires are an essential part of the savannah ecosystem), and illegal hunting of wildlife species (black rhinoceros were extirpated of Nigeria just few decades ago) have also contributed significantly for the loss of biodiversity (Nicolas et al. 2010; Chidumayo et al. 2011; WWF 2013g).

Nowadays the main patches of natural habitat are located in protected areas around all the countries in the region, and represent about 6.7% of the total area (WWF 2013g). Protected areas include forest reserves like the “Projet d’Aménagement des Massifs Forestiers d’Agoua, des Monts Kouffé et de Wari-Marô” in Benin (Nicolas et al. 2010), and many national parks such as Boucle du Baoulé in Mali, the River Gambia in Gambia, Niokolo-Koba in Senegal, Kainji Lake in Nigeria, Mole in Ghana, and Comoe in Côte d’Ivoire (WWF 2013g). Another relevant protected area is the W national park that includes areas of three states, Burkina-Faso, Benin and Niger (Baudron et al. 2009). Moreover, the region encompasses various Ramsar sites (Ramsar 2013) and IBAs (BirdLife International 2013).

3.1.4 South Africa

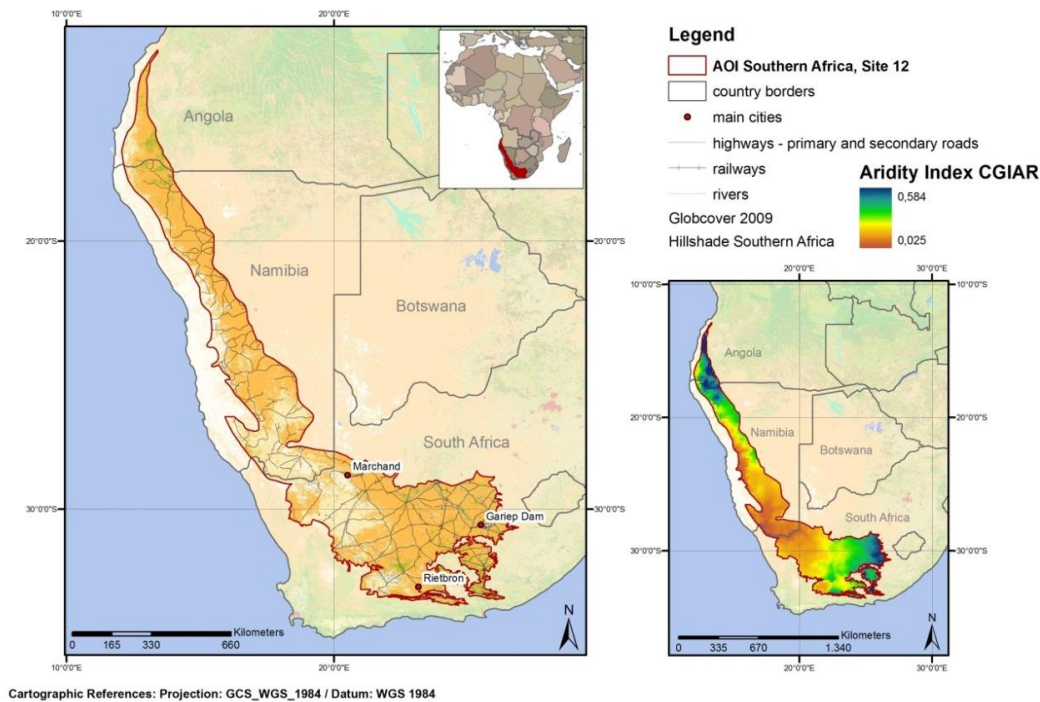


Figure 44: Dryland test site – South Africa.

The study area is comprised of two ecoregions (Figure 44): the Nama Karoo Biome consists of a vast and open arid region, characterized by dwarf shrubland vegetation, the majority of which is located on the central plateau of the Cape Province in South Africa, but also extends further northwest into Namibia (Palmer & Hoffman 1997; WWF 2013h); and the Namibian savannah woodland that stretches from western Angola down into Namibia, ending just north of the Groot Karas Berg plateau, forming a narrow belt that covers the Great Escarpment and delimiting the Namib desert in the west (WWF 2013i). The topography of the region is very diverse, encompassing many plateaus and mountains, including the highest peak in Namibia, the Brandberg (2038 m) (WWF 2013i). Typical landscape features are the mesas (flat-topped isolated mountains) that act as inselbergs, providing more suitable climatic and ecological conditions for more water dependent species, in comparison with the arid settings of the surrounding areas (Burke 2003). These mesas are normally formed of dolerite that is more resistant to erosion than the predominant limestone, sandstone and shale (WWF 2013hi). The region is subjected to a harsh and seasonal climate, with low unpredictable rainfalls concentrated mostly during the summer (up to 70% during January-March), and high daily fluctuations of temperature (Chase et al. 2010; WWF 2013hi). The precipitation displays an increasing gradient from west to east and from north to south. In the woodland savannah, mean annual rainfall ranges between 50 in 200 mm from west to east, while in the Nama Karoo region mean annual rainfall is generally higher in the northern part (500 mm) and decreases to the south (around 100 mm) (Palmer & Hoffmann 1997; WWF 2013hi). However, as mentioned before, high elevation landscapes features like plateaus, mesas and mountains can hold more humid conditions despite the more dry surroundings (Burke et al. 2003). In the more northern regions, the Benguela Current (Atlantic Ocean) brings a fresh sea breeze, so temperatures are more moderate during the summer and in winter frost rarely occurs. In more

southern and continental areas that do not receive the cooling effect of the current, temperature variations becomes more extreme and frost is common, with mean monthly temperatures going above zero during winter, and the mean maximum monthly temperature surpassing 40°C in the summer (Burke et al. 2003; WWF 2013hi).

The diversity of vegetation in the study area reflects the two biomes, the variety of landscape features and associated soils and microclimates. In the north, the savannah is dominated by the mopane (*Colophospermum mopane*) that can occur as a tree and form dense woodlands, or as short-stemmed shrub amongst other trees like *Balanites welitschii* and of the genus *Sesamothamnus* (Mags et al. 1998; WWF 2013i). As we go southwards and reach the Brandberg Mountain, the vegetation reaches a transition zone between savannah and semi-desert, demonstrating great diversity and endemism. Typical species of this zone include *Euphorbia guerichiana*, *Cyphostemma* spp., the quiver tree (*Aloe dichotoma*), *Comiphora* spp., and two endemic species of *Acacia*, *A. montisustii* and *A. robynsiana* (Mags et al. 1998; WWF 2013i). To the south begins the Nama-Karoo biome and its representative dwarf shrubs and grasses. These include shrub species of the genus *Drosanthemum*, *Eriocephalus*, *Galenia*, *Rhigozum* and *Ruschia*, while for grasses we have the cases of *Aristida*, *Digitaria*, *Enneapogon* and *Stipagrostis*. Trees like *Acacia karoo*, *Diospyros lycioides* and *Tamarix usneoides* are more restricted to watercourses (Palmer & Hoffman 1997; WWF 2013h).

In terms of fauna, the Namibian woodland savannah presents a higher species richness and endemism than the poorer Nama-Karoo, especially in the mountainous region of Brandberg (Simmons et al. 1998; Proches & Cowling 2006). The first biome holds two endemic amphibians, the Okahandja toad (*Bufo hoeschi*) and the Mossamedes toad (*B. grandisonae*), while for reptiles we have endemic or near-endemic species like the Albert's burrowing skink (*Sepsina alberti*), the Nama padloper tortoise (*Homopus solus*), two lizards of the genus *Corydillus* (*C. namaquensis* and *C. pustulatus*) and the Brandberg thick-toed gecko (*Pachydactylus gaiasensis*) (Simmons et al. 1998; WWF 2013i). For the mammalian fauna, the woodland savannah harbours important populations of elephant (*Loxodonta africana*), black rhinoceros (*Diceros bicornis*), lion (*Panthera leo*) and cheetah (*Acinonyx jubatus*), among other large mammals. Endemic species are mostly smaller species like the Angola wing-gland bat (*Myotis seabrai*), slender mongoose (*Galerella swalius*) and rock mouse (*Petromyscus shortridgei*). The Kaoko Escarpment has the highest bird diversity, hosting up to 297 species like the greybacked cisticola (*Cisticola subruficapillus*), the Cinderella waxbill (*Estrilda thomensis*) and the Herero chat (*Namibornis herero*) (Simmons et al. 1998; WWF 2013i).

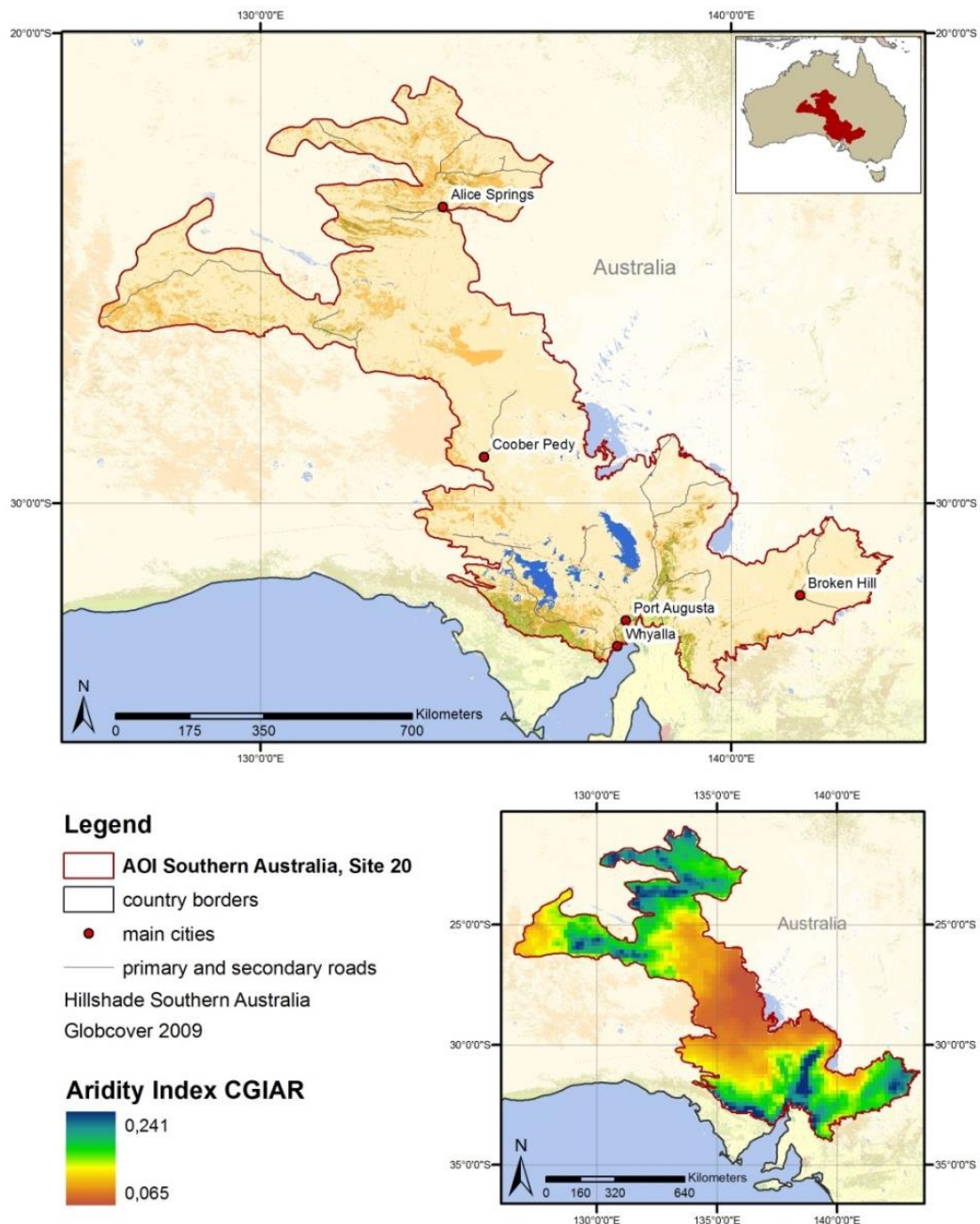
Relatively to the Karoo, endemism is lower since most species come from adjacent regions. Examples of herpetofauna include the olive toad (*Bufo garmani*), Karoo dwarf chameleon (*Bradypodion karrooicum*) and Boulenger's padloper tortoise (*Homopus boulengeri*). Amongst mammals, one of the rarest and most endangered species in the world occurs in the region, the riverine rabbit (*Bunolagus monticularis*), as well as the Grant's rock mouse (*Aethomys granti*), bushy-tailed hairy-footed gerbil (*Gerbillurus vallonius*), bat-eared fox (*Otocyon megalotis*) and Cape fox (*Vulpes chama*). Finally for the case of avian fauna, relevant examples comprise of the Karoo (*Cercomela sclegelii*) and tractrac chats (*C. tractrac*), tawny (*Aquilarapaz*) and martial eagles (*Polemaetus bellicosus*), red lark (*Certhilauda burra*) and Karoo scrub robin (*Cercotrichas coryphaeus*) (Vernon 1999; WWF 2013h).

The major threats to biodiversity are more or less common to both biomes. Heavy agriculture grazing, deforestation and mining activities have dire consequences, destroying

habitats and degrading the soils (Fahse et al. 1998; Hoffmann & Zeller 2005; du Toit et al. 2011; WWF 2013hi). In Namibia during the first half of the century, severe trophy hunting of large mammals resulted in major range constrictions, more than 95% over the last 200 years for the cases of Plains zebra (*Equus burchelli*) and lion (Griffin 1998). The introduction of exotic species is also an issue, and in Namibia problems exist related with cross-breeding between domestic cat (*Felis domesticus*) and wild cats (*F. sylvestris* and *F. nigripes*) (Griffin 1998), while in the Nama-Karoo exotic plants species can outcompete native plants (WWF 2013h). Climate change will undoubtedly cause changes in biodiversity with reduce rainfalls and longer drought periods, which can for example influence the ecology of birds (Altwegg & Anderson 2009).

Around 14% of Namibia is designated as conservation area, to which you have to add additional game reserves, private and NGO funded protected areas (Barnard et al. 1998). Fortunately these areas seem to hold most of the richest endemicity regions, such as the Kaoko Escarpment and Brandberg Mountain (Simmons et al. 1998). However, both the woodland savannah and Karoo biomes still seem slightly underrepresented within protected areas (Barnard et al. 1998). Important examples of protected regions include the Etosha National Park in Namibia which was once the largest nature reserve in the world, the Brandberg National Monument, and the Karoo National Park in South Africa (Barnard et al. 1998; Rubin et al. 2001). Most of these conservation areas are also included in the Ramsar site list (Ramsar 2013) and Important Bird Areas (BirdLife International 2013).

3.1.5 Australien



Cartographic References: Projection: GCS_WGS_1984 / Datum: WGS 1984

Figure 45: Dryland test site – Australien.

Under the Interim Biogeographic Regionalisation for Australia (IBRA) planning framework, an important part of the Australia's Strategy for the National Reserve System 2009-2030, the country's landscape has been divided into 89 bioregions (NRMMC 2009; IBRA 2012). The study area (Figure 45) spans through four states, mostly in South Australia and Northern Territory but also smaller parts of New South Wales and Western Australia. The study area encompasses nine bioregions (from south to north): Flinders Lofty Block (FLB),

Gawler (GAW), Broken Hill Complex (BHC), Stony Plains (STP), Simpson-Strzelecki Dunefields (SSD), Finke (FIN), Central Ranges (CER), MacDonnell Ranges (MAC) and Burt Plain (BRT) (Bastin & ACRIS 2008a,b,c,d,e,f,g,h,i). All these bioregions can be considered parts of the Tirari-Sturt Stony Desert. The combined area of the nine bioregions (and thus of the study area approximately) is of 933,550 km², with the SSD being the largest (272,920 km²) and the MAC the smallest (39,290 km²) (Bastin & ACRIS 2008a,b,c,d,e,f,g,h,i). All the bioregions are within the desert and xeric shrubland ecoregion of Australia, with predominant vegetation types being grass and shrubland, mulga (acacia) woodland and eucalyptus trees on hills and areas with higher rainfall (Bastin & ACRIS 2008a,b,c,d,e,f,g,h,i). The study area is subjected to a semiarid to arid climate, with sparse and unreliable rainfalls that tend to follow a decreasing gradient from south to north, ranging from 217 mm in FLB to 118 mm in STP (spatially averaged medians from 1890-2005), and high evapo-transpiration rates. However, the landscape can also influence this pattern and for example rainfall in the MAC and BRT bioregions are of 228 and 243 mm, respectively, due to mountain ranges. Precipitation in the southern regions occurs mainly during winter, while summer storms dominate in the more northern reaches. Mean maximum temperatures during the summer range from 18 to 38°C, but can reach up to 50°C in the more central and arid regions (SSD and CER for example), while during winter it's common for temperature to range from 5 to 20°C (Bastin & ACRIS 2008a,b,c,d,e,f,g,h,i; ANRA 2013 a,b,c,d,e,f,g,h,i).

As one can imagine, the topography is very diverse, with successive mountain ranges, rocky hills, ridges, wide flat plains, sand dunes, salt lakes, watercourses, alluvial plains and springs. The geological background comprises mainly very old formations (Proterozoic and Palaeozoic) of metamorphosed sedimentary and igneous rock, among other later topographic features that span the entire geological time scale (ANRA 2013 a,b,c,d,e,f,g,h,i).

The study area holds an impressive biological diversity despite the harsh conditions. In terms of flora, all bioregions combined may hold over 2000 taxa, including many endemics and threatened species. The vegetation follows closely the topographic and climatic diversity. Within the FLB region's relatively more humid hills is common to find many species of eucalyptus (*Eucalyptus* spp.). As we move to more arid and flat regions the dominant types of vegetation become the mulga, chenopod and mallee woodlands with many acacias (*Acacia* spp.), grasses (*Astrelba* spp., *Danthonia* spp., *Stipa* spp., among others) and shrubs (*Atriplex* spp., *Maireana* spp., *Senna* spp., etc.) (Bastin & ACRIS 2008a,b,c,d,e,f,g,h,i; ANRA 2013 a,b,c,d,e,f,g,h,i).

The fauna in the region also presents a great species richness and endemism, with around 20 amphibians, over 100 reptiles, up to 50 mammals and over 200 birds. Relevant herpetofauna examples include the stream bank froglet (*Crinia riparia*), two agamid lizards (*Ctenophorus rufescens* and *C. maculosus*), a skink (*Lerista speciosa*) and a dtella or web-toed gecko (*Gehyra variegata*). Important mammal species include the yellow-toed (*Petrogale xanthopus*) and black-footed rock wallabies (*P. lateralis*), ghost bat (*Macroderma gigas*), dingo (*Canis lupus dingo*) and red kangaroo (*Macropus rufus*), among others, but common species are also the introduced European rabbit (*Oryctolagus cuniculus*) and red fox (*Vulpes vulpes*), besides feral domestic species like goats (*Capri hircus*), camels (*Camelus dromedarius*) and cats (*Felis cattus*). In relation to birds, key species include the Australian bustard (*Ardeotis australis*), the eyrean grasswren (*Amytornis goyderi*), scarlet-chested parrot (*Neophema splendida*), rufous-crowned emu-wren (*Stipiturus ruficeps*), grey falcon (*Falco hypoleucus*) and



Wedge-tailed eagle (*Aquila audax*) (Bastin & ACRIS 2008a,b,c,d,e,f,g,h,i; ANRA 2013 a,b,c,d,e,f,g,h,i).

The main concerns in the study area are related with overgrazing and pest species. The region has been affected by intense pastoralism, especially by sheep and cattle. Rabbits and goats are also responsible for overgrazing, which exposes soils to erosion, reduces the recruitment rate of native plants and reduces the quality of habitat for native animals (DEH 2009; ANRA 2013 a,b,c,d,e,f,g,h,i). Many exotic plants are also present ,up to 150, such as buffel (*Cenchrus ciliaris*) and couch grasses (*Cynodon dactylon*), mimosa bush (*Acacia farnesiana*), Bathurst burr (*Xanthium spinosum*) and athel pine (*Tamarix aphylla*) (ANRA 2013 a,b,c,d,e,f,g,h,i). The region also harbours mining explorations that can seriously transform and degrade the land (ANRA 2013 a,b,c,d,e,f,g,h,i). If we take into account the already arid climate, future climatic changes and the intense overgrazing, alteration and increase of severity of fire regimes poses also a very significant threat (Russel-Smith et al. 2003; DEH 2009).

Besides pastoral, agricultural and mining leases, the region also holds many extents of Aboriginal land, which is under less human pressure (Russel-Smith et al. 2003). Some bioregions are better covered by conservation reserves than others, but still most of the important biodiversity (as well as topographic) features are well represented, and many National Parks (NP) and reserves have been established, like the Sturt NP, Flinders Ranges NP, Lake Gairdner NP, Mootwingee NP, Simpson Desert Conservation Park and Innamincka Regional Reserve (ANRA 2013 a,b,c,d,e,f,g,h,i). Some IBAs are also located within the region (BirdLife International 2013).

3.2 Validation of Indicators with Biodiversity Data

Biodiversity distribution data is a fundamental aspect of the validation process of DIVERSITY II products. Such distribution data is essential to determine the accuracy and usefulness of derived biodiversity indicators produced within the scope of the project. As such, distribution data was collected from multiple sources that include: GBIF, Global Biodiversity Facility Information (www.gbif.org); Distribution Atlases of biodiversity at national or continental level; and unpublished data available at CIBIO/UP from particular regions (e.g., S-Europe or West Sudanian Savannah). The Global Biodiversity Information Facility (GBIF) is an international open data infrastructure. It has developed a worldwide information network through which natural history museums and other institutions publish their databases (Guralnick et al 2007). Since its foundation, it has published over 370 million records of species (Costello et al 2013). Despite the amount of data available, there are some concerns about accuracy and quality of the geographic, temporal, and taxonomic coverage (Guralnick et al 2007, Costello et al 2013). Nevertheless, GBIF developed strategies and action plans towards expedited discovery and publishing of primary biodiversity data and for frequent monitoring, and gap analyses (GBIF 2010).

The following chapters describe for both status and trend analyses the available data sources, methodologies used and results. These included: collection of biodiversity distribution data and environmental data for each test site; development of species distribution models; combination of models to derive predictions of species richness; and correlating biodiversity indicators with species richness predictions. Results of predicted species richness for the five dryland test sites are detailed first, followed by correlation outputs.

3.2.1 Status

Available distribution data for drylands comprise observations of terrestrial non-volant vertebrates: amphibians, reptiles and mammals. These groups have close relationships with

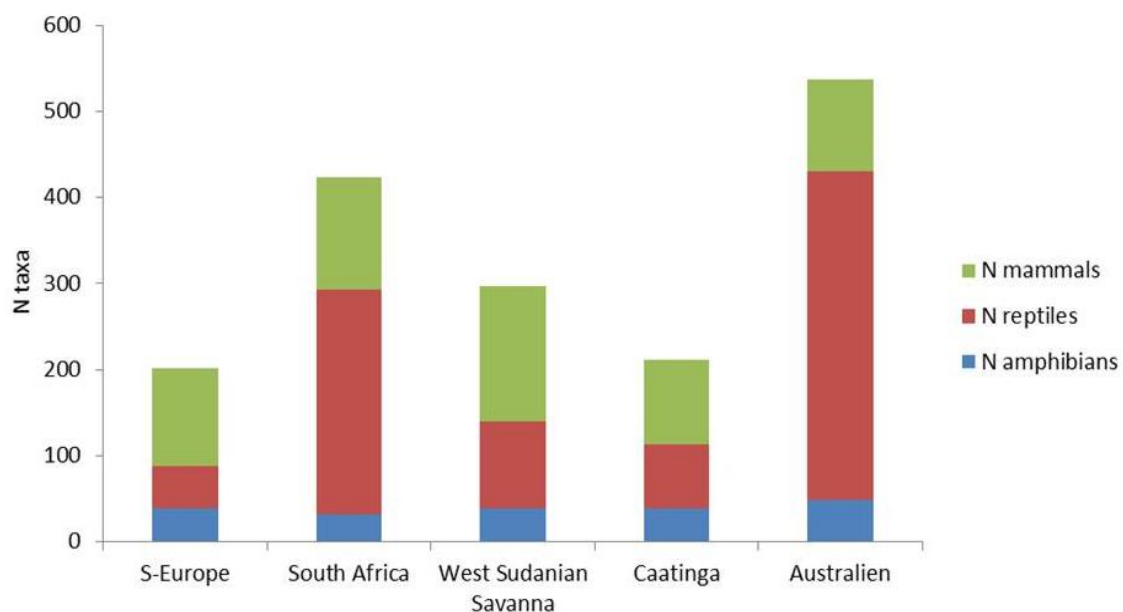


Figure 46: Number of terrestrial non-volant vertebrate taxa for each dryland test site.

the environmental parameters to be estimated during the project (Lomolino et al. 2010). The intersection between the five drylands and the databases identified a total of 1670 taxa. The South Africa and Australien drylands are particularly species-rich, holding each more than 400 species, and reptiles are the group contributing mostly to such increased diversity (Figure 46). The remaining test sites have similar diversity numbers (200 to 300 species each) and display similar proportions among each of the three vertebrates under study.

Validation of biodiversity indicators for drylands with *in-situ* biodiversity data is aimed to detect direct relationships between biodiversity indicators and species richness. The validation workflow follows four steps (Figure 47), which include: collection of biodiversity distribution data and environmental data for each test site; development of species distribution models; combination of models to derive predictions of species richness; and correlating biodiversity indicators with species richness predictions. These four steps are detailed in the following subsections.

3.2.1.1 Biodiversity and Environmental Data

Biodiversity distribution data at 1x1 km of resolution was obtained from GBIF (www.gbif.org), and complemented in some test-sites with unpublished data available at CIBIO/UP (e.g., S-Europe or West Sudanian Savannah). Nearly 400,000 presence records from 1670 species have been collected and are available as ArcGIS shapefiles. However, quality issues have been raised when using available data at GBIF, namely related with outdated taxonomic identification of species and other taxa (most of the times records come from old museum collections), and most importantly, spatially biased distribution data due to unbalanced sampling, data storage and availability (Beck et al. 2013). To surpass this issue, a taxonomic review of selected taxa was carried out based on the IUCN Red List of Threatened Species (www.iucnredlist.org), and additionally for reptiles The Reptile Database (reptile-database.reptarium.cz). Afterwards, presence records were checked for data duplication (same species records within one pixel), and any taxa with a number of records below 15 was removed before analyses.

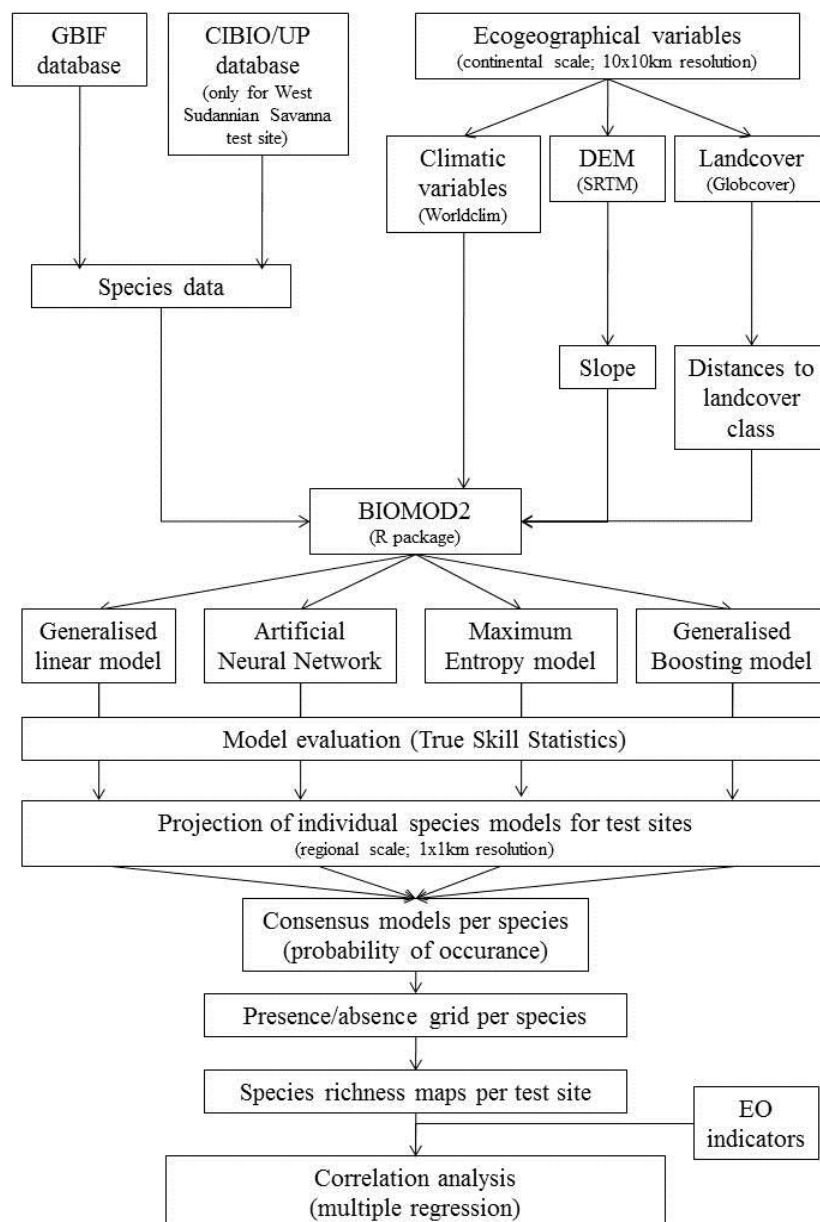


Figure 47: Validation protocol for drylands based on *in-situ* biodiversity data.

Three sets of environmental factors, or ecogeographical variables (hereafter EGV), were used for building the ecological models. These sets (Table 10) include: 1) one topographical grid (USGS, 2006) that was used to derive the variable Slope; 2) nineteen climate grids representing averages, extremes and seasonality in precipitation and temperature levels (Hijmans et al., 2005); and 3) a land-cover grid from the year 2009 (ESA 2010). To reduce the correlation between the climatic grids and the number of variables, only six of the nineteen initial climatic grids were used for modelling proposes (BIO1, BIO5, BIO6, BIO12, BIO13 and BIO14). In order to convert the categorical land-cover EGV into a continuous variable, one binary grid was created for each habitat type that covered more than 5% of each continent (Table 11). The Euclidean distance of each grid cell to the closest habitat type was calculated for each individual habitat grid. Finally, two sets of EGVs were created: 1) a continental set of

EGVs at 10x10km; and 2) a regional set of EGVs at 1x1km corresponding to each test site. All EGVs were resampled from the original to 10x10km and 1x1km, respectively.

Table 10: Environmental factors considered for developing species distribution models for each target species in each dryland test-site.

CODE	Factor	Source
BIO1	Annual Mean Temperature	Worldclim (http://www.worldclim.org/)
BIO2	Mean Diurnal Range (Mean of monthly (max temp - min temp))	Worldclim (http://www.worldclim.org/)
BIO3	Isothermality (BIO2/BIO7) (* 100)	Worldclim (http://www.worldclim.org/)
BIO4	Temperature Seasonality (standard deviation *100)	Worldclim (http://www.worldclim.org/)
BIO5	Max Temperature of Warmest Month	Worldclim (http://www.worldclim.org/)
BIO6	Min Temperature of Coldest Month	Worldclim (http://www.worldclim.org/)
BIO7	Temperature Annual Range (BIO5-BIO6)	Worldclim (http://www.worldclim.org/)
BIO8	Mean Temperature of Wettest Quarter	Worldclim (http://www.worldclim.org/)
BIO9	Mean Temperature of Driest Quarter	Worldclim (http://www.worldclim.org/)
BIO10	Mean Temperature of Warmest Quarter	Worldclim (http://www.worldclim.org/)
BIO11	Mean Temperature of Coldest Quarter	Worldclim (http://www.worldclim.org/)
BIO12	Annual Precipitation	Worldclim (http://www.worldclim.org/)
BIO13	Precipitation of Wettest Month	Worldclim (http://www.worldclim.org/)
BIO14	Precipitation of Driest Month	Worldclim (http://www.worldclim.org/)
BIO15	Precipitation Seasonality (Coefficient of Variation)	Worldclim (http://www.worldclim.org/)
BIO16	Precipitation of Wettest Quarter	Worldclim (http://www.worldclim.org/)
BIO17	Precipitation of Driest Quarter	Worldclim (http://www.worldclim.org/)
BIO18	Precipitation of Warmest Quarter	Worldclim (http://www.worldclim.org/)
BIO19	Precipitation of Coldest Quarter	Worldclim (http://www.worldclim.org/)
Slope	Slope (to be derived from a DEM)	SRTM (http://srtm.usgs.gov/index.html)
Land-cover	Land-cover types	Globcover (http://due.esrin.esa.int/globcover/)

Table 11: Land cover categories by continent used for developing species distribution models.

Continent	Code	Description
Africa	D03	Mosaic vegetation (50-70%) / cropland (20-50%)
	D07	Open (15-40%) broadleaved deciduous forest/woodland (>5m)
	D09	Closed to open (>15%) (broadleaved or needleleaved, evergreen or deciduous) shrubland (<5m)
	D10	Closed to open (>15%) herbaceous vegetation (grassland, savannahs or lichens/mosses)
	D14	Bare areas
	D15	Consolidated bare areas (hardpans, gravels, bare rock, stones, boulders)
	D16	Non-consolidated bare areas (sandy desert)

Australia	D03	Closed to open (>15%) broadleaved evergreen or semi-deciduous forest (>5m)
	D05	Mosaic forest or shrubland (50-70%) / grassland (20-50%)
	D06	Mosaic grassland (50-70%) / forest or shrubland (20-50%)
	D01	Rainfed croplands
	D07	Closed to open (>15%) (broadleaved or needleleaved, evergreen or deciduous) shrubland (<5m)
	D08	Closed to open (>15%) herbaceous vegetation (grassland, savannahs or lichens/mosses)
	D09	Sparse (<15%) vegetation
Europe	D01	Rainfed croplands / herbaceous crops / shrub or tree crops
	D02	Mosaic cropland (50-70%) / vegetation (grassland/shrubland/forest) (20-50%)
	D04	Closed (>40%) broadleaved deciduous forest (>5m)
	D06	Open (15-40%) needleleaved deciduous or evergreen forest (>5m)
	D08	Closed to open (>15%) mixed broadleaved and needleleaved forest (>5m)
	D12	Sparse (<15%) vegetation (grassland or shrubland)
South America	D01	Rainfed croplands and shrub or tree crops
	D02	Mosaic cropland (50-70%) / grassland/shrubland/forest (20-50%)
	D04	Closed to open (>15%) broadleaved evergreen or semi-deciduous forest (>5m)
	D03	Mosaic vegetation (grassland/shrubland/forest) (50-70%) / cropland (20-50%)
	D10	Closed to open (>15%) (broadleaved or needleleaved, evergreen or deciduous) shrubland (<5m)
	D05	Closed (>40%) broadleaved evergreen and/or semi-deciduous forest (>5m)

3.2.1.2 Species Distribution Models and Species Richness

Species distribution models (SDM) have become essential tools in biodiversity conservation and management (Guisan et al., 2007, Vale et al., 2014). Statistically, they relate the geographical distribution of species or communities to present environment, and their combination with Geographical Information Systems (GIS) and remote sensing has prompted conservation biology studies with more robust analytical methods (Guisan & Zimmermann 2000; Elith & Leathwick, 2009; Cord et al. 2013). SDMs are useful tools to analyse geographic-related processes for conservation planning, such as the identification of suitable habitat areas for rare species (e.g., Torres et al. 2010; Brito et al. 2011a) or over large and remote study areas (Bruto et al. 2009), the estimation of population size for peripheral populations (e.g., Santos et al. 2009), the prediction of climate-change induced range shifts (e.g., Carvalho et al. 2010; Brito et al. 2011b), the assessment of conservation status of poorly known species (e.g., Vale et al. 2012a,b), and the design of reserves (e.g., Carvalho et al. 2011a,b; Vasconcelos et al. 2012). In the context of the DIVERSITY II project, SDMs were used to determine suitable areas for the occurrence of the selected vertebrates in the dryland test-sites, which were used to derive potential species richness per grid cell.

A set of 10 modelling techniques to predict the distribution of each of the target species were available to use:

- Maximum Entropy model (MAXENT) is a machine-learning method that estimates the range of a species with the constraint that the expected value of each variable (or its transform and/or interactions) should match its empirical average, *i.e.* the average

value for a set of sample points taken from the species-target distribution (Phillips et al. 2004, 2006, 2008). It is a SDM that consistently performs well in comparison to other methods, especially at low samples sizes (Elith et al. 2006; Hernandez et al. 2006; Hijmans & Graham 2006; Wisz et al. 2008);

- Generalised Linear models (GLM), which are generalization of ordinary linear regression that allows for response variables that have other than a normal distribution (Nelder & Wedderburn 1972). GLMs generalize linear regression by allowing the linear model to be related to the response variable (species presence) via a link function and by allowing the magnitude of the variance of each measurement to be a function of its predicted value;
- Generalised Additive models (GAM), which blend properties of GLMs with additive models, i.e. models based in nonparametric regression methods (Hastie & Tibshirani 1990);
- Classification Tree Analysis (CTA) is a type of machine learning algorithm based on classification trees that are used to predict membership of cases or objects in the classes of a categorical dependent variable (species presence) from their measurements on one or more predictor variables (Ripley 1996);
- Artificial Neural Networks (ANN) are machine-learning methods, inspired by biological neural networks, which are used for modelling complex relationships between inputs and outputs (species presence) or for find patterns in data (Lek et al. 1996);
- Generalised Boosting models (GBM), which apply boosting methods to regression trees (Friedman 1999a,b);
- Breiman and Cutler's random forest for classification and regression (RF), are an ensemble learning method for classification (and regression) that operate by constructing a multitude of decision trees at training time and outputting the class that is the mode of the classes output by individual trees (Breiman 2001);
- Flexible Discriminant Analysis (FDA) is a method for classification (supervised) based on mixture models that correspond to an extension of linear discriminant analysis, where the mixture of models is used to obtain density estimation for each class (Hastie & Tibshirani 1996);
- Multiple Adaptive Regression Splines (MARS), which is a form of non-parametric regression technique that can be seen as an extension of linear models that automatically models non-linearities and interactions between variables (Friedman 1991);
- Surface Range Envelop (SRE) is an algorithm to compute the similarity of a location by comparing the values of environmental variables at any location to a percentile distribution of the values at known locations of occurrence (Nix 1986, Busby 1991, McMahon et al. 1996).

Fitting SDMs involves numerous choices and well-justified decisions (Guisan & Thuiller, 2005; Buisson et al., 2010; Beale & Lennon, 2012). The study area extent and resolution (pixel size) are two of the most important factors affecting SDMs predictions (Guisan et al., 2007, Vale et al., 2014). For instance, the choice of the pixel size for modelling could be related to the cell size of available environmental data (e.g. the bioclimatic variables are available at 1km²), characteristics of the species data (e.g. geographical accuracy) or finally, the computer

power (i.e. too many pixels may require too demanding computer resources). The pixel size is also related with ecological and management issues. Different pixel sizes can influence the perception of a phenomenon, such as patterns of presence or abundance or affect the relevance of the output for management applications (Guisan et al., 2007, Araújo et al., 2005). Generally, for regional and local assessments, regional models at high resolution data outperform continental models built with coarse resolution data (Vale et al., 2014). However, the limited availability of high resolution data can sometimes preclude its use, and in this context, downscaled models may constitute an alternative way to identify suitable areas for conservation actions and areas where local distribution data cannot be achieved (Araújo et al., 2005; Barbosa et al., 2010, Vale et al., 2014). For instance, Barbosa and colleagues (2010) analysed the performance of downscaled models, built from occurrence data at 10×10 km resolution, and projected to a resolution 100 times finer (1×1 km). Their results suggested downscaled predictions were significantly related to the observed presence and absence of species. The predictive ability of coarse-resolution and downscaled models are generally consistent (Barbosa et al., 2010). Moreover, downscaled distributions could be also able to reveal environmental gradients otherwise blurred by coarse-resolution data (Araújo et al., 2005). As such, in the absence of high resolution data or computation power to deal with high amounts of high resolution data, downscaling is a valid alternative method as they can still capture general gradients in environmental quality and perform satisfactorily at predicting species occurrence at regional scale and at finer resolutions.

The SDMs were built at a 10×10 km resolution using the continental set of EGVs and then projected to the test sites at 1×1 km. All SDMs were performed on Biomod2 package (<http://cran.r-project.org/web/packages/biomod2/index.html>) (Thuiller et al. 2009, Thuiller et al. 2012). Biomod2 is the updated object-oriented version of BIOMOD package, which is a collection of functions running within the R software v. 3.0.1 (R Core Team 2013) for ensemble forecasting of species distributions, projecting model predictions to different spatial/temporal environmental conditions and a summary of model statistics is described in Thuiller et al. (2009). After a preliminary analysis, from the initial 10 modelling algorithms available, only four (GLM, ANN, GBM and MaxEnt) were chosen as they had high performances for all datasets. Indeed, these four modelling algorithms were reported to have high performances and with relatively low sample sizes (e.g., Elith et al. 2006; Hernandez et al. 2006).

The four different modelling algorithms require distinct observational data: both presence and absence data for GLM, and presence-only data in ANN, GBM, and MaxEnt (Phillips et al. 2006; Thuiller et al. 2009, 2012). Continental EGVs (10×10 km) and the presence records were imported to Biomod2. Two datasets of pseudo-absences were randomly created for each species in Biomod2. A large number of pseudo-absences were created for each dataset of each species (10 000) to obtain the most accurate results reported using regression and machine learning algorithms (Barbet-Massin et al. 2012).

All models were produced using default Biomod2 parameters whenever possible (Thuiller et al. 2009). The number of pseudo-absences were chosen to have the same weight than presence data in the calibration process of the models (i.e., prevalence=0.5) (Barbet-Massin et al. 2012). Biomod2 allows evaluation of model performance on different data split runs and then allows using 100% of the data to make a final calibration of the predictive models. Thus, 80% of occurrence data were randomly assigned to model training with the remaining 20% for model testing. Individual models were evaluated using the true skill

statistics (TSS), which has been demonstrated as highly effective for testing the performance of models (see Allouche et al. 2006). Only models with $TSS > 0.70$ were kept for subsequent analyses, with the exception of the Caatinga test site, for which the value was decreased to 0.5. The values were chosen after analyses of models output and resulted from a compromise between performance and replicates per species (Martínez-Freiría et al. 2013).

In the context of the DIVERSITY II project and to determine suitable areas for the vertebrates in the dryland test-sites, individual model replicates were projected to the test sites (regional EGVs at 1x1km). The downscaling method was chosen due to a combination of factors. First, SDMs were built a continental scale to include, as much as possible, the complete species' environmental range. The used of restricted data reduces the combinations of environmental conditions under which the models are calibrated (Thuiller et al. 2004). Second, given that models were calibrated at continental scale, 10x10km resolution was chosen due to software limitations but also, to reduce the impact of some data from GBIF with low geographical accuracy. Finally, models were projected to study sites at finer resolution (1x1km), which is the original pixel size of the bioclimatic variables. As such and given the high amount of data, the downscale approach was the strategy most feasible and simultaneously accurate to identify species' suitable areas for the DIVERSITY II project. The individual model replicates projected to the test sites were added to generate mean forecasts of probability of species presence in each test site (Araújo & New 2007; Marmion et al. 2009). To obtain species richness maps for each test site, the consensus forecasted probabilities of occurrence were converted to a binary value of predicted presence/absence using an optimized threshold automatically selected by Biomod2 in order to maximise both the sensitivity and specificity of models, i.e. to maximise the correct classification rate of both observed presence and absence data (Liu et al. 2005, 2013). Finally, species richness was estimated by adding the individual predictions of species presence/absence in each grid cell (Araújo and Williams 2000, Cumming 2000, Soares and Brito 2007, Benito et al 2013).

3.2.1.3 Correlating Biodiversity Indicators with Species Richness

Biodiversity indicators derived by DIVERSITY II were contrasted against predicted species richness. A total of 27 indicators were used (including mean and variation): 1) fAPAR cyclic fraction, dry season and vegetation year; 2) Rainfall cyclic fraction and vegetation year 3) Rain Use Efficiency (RUE) cyclic fraction, dry season and vegetation year; 4) Soil Moisture cyclic fraction and vegetation year; 5) Soil Moisture Use Efficiency (SMUE) cyclic fraction, dry season and vegetation year; and 6) Length of Vegetation Season.

Correlation coefficients between biodiversity indicators and species richness were then calculated using the "Band Collection Statistics" tool in ArcGIS 10.1. Significant correlations ($r > 0.50$, absolute value) should provide evidence of consistencies between derived products and biodiversity data across diverse environmental backgrounds (considering that the correlations follow a linear distribution).

To assess if other explanatory models were better suited (logarithmic and polynomial functions, among others), values (above zero) of both indicators and species richness for each taxonomic group were extracted based on a set of 1000 points randomly generated in ArcGIS 10.1, which were subsequently plotted. Analyses were not done for total species richness to avoid confounding effects between distinct group responses to the same indicator.

3.2.1.4 Results

1) Species richness

Species richness maps for the five drylands test site are accompanied by the respective aridity index map for visual comparison.

Despite the number of taxa in the Caatinga test site (Figure 48) being similar to those of the Southern Europe test site (Figure 46), the number of available GBIF records was considerably lower (N=4954 and N=282406, respectively). Subsequently, many areas retrieved zero species presence mostly due to lack of data, despite multiple TSS threshold essays for model acceptance (as explained above). Nevertheless, some patterns are discernible and are concordant between the three taxonomic groups. Species richness seems to be higher in the eastern limits of the test site, most likely because of the proximity with other rich biomes such as the Atlantic forest and the Cerrado.

Figure 49 illustrates the results for the Southern Europe test site. Every pixel included at least one species presence. Species richness patterns are not clearly concordant between amphibians, reptiles and mammals, but overall the areas with highest richness include central Iberian Peninsula, North eastern Spain, southern France, and island of Sicily.

For the West Sudanian test site (Figure 50), many areas are depicted with zero species (except reptiles), which could be related with lack of distribution data as in the Caatinga test site. However, the differences found are also related with the different biological traits that characterize each group, especially for amphibians that are very dependent in moister (Lomolino et al. 2010). Species richness seems to be higher along some areas of the southern and northern limits of the test site. The high reptile species richness observed in North western extension of the test site could be related with a more recent intensive sampling effort in the region (CIBIO/UP unpublished data).

The South Africa test site (Figure 51) was the one presenting the highest species richness. Reptile richness is considerably high in the central-northern area, while both amphibians and mammal species are more numerous in southern Africa. These patterns result in relatively high total species richness throughout the entire test site, with 30 as the minimum number of species per pixel.

Finally, the Australien test site (Figure 52) reveals very similar patterns across each taxonomic group, with higher levels of species richness being found in the northern and southern limits of the test sites, whereas the most central areas are species poor. Still, although not visible in the maps, some pixels within the central area also present relatively high diversity, which could be related with desert water springs found in the region (Murphy et al. 2012).

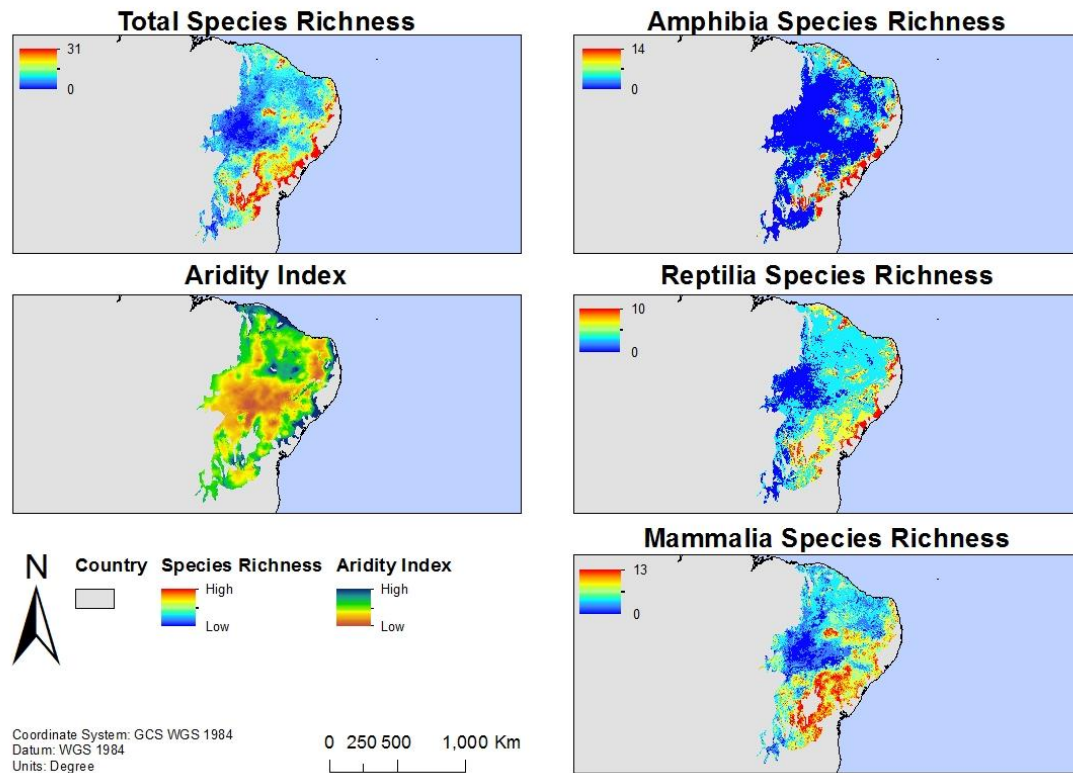


Figure 48: Species richness and aridity index for the Caatinga test site.

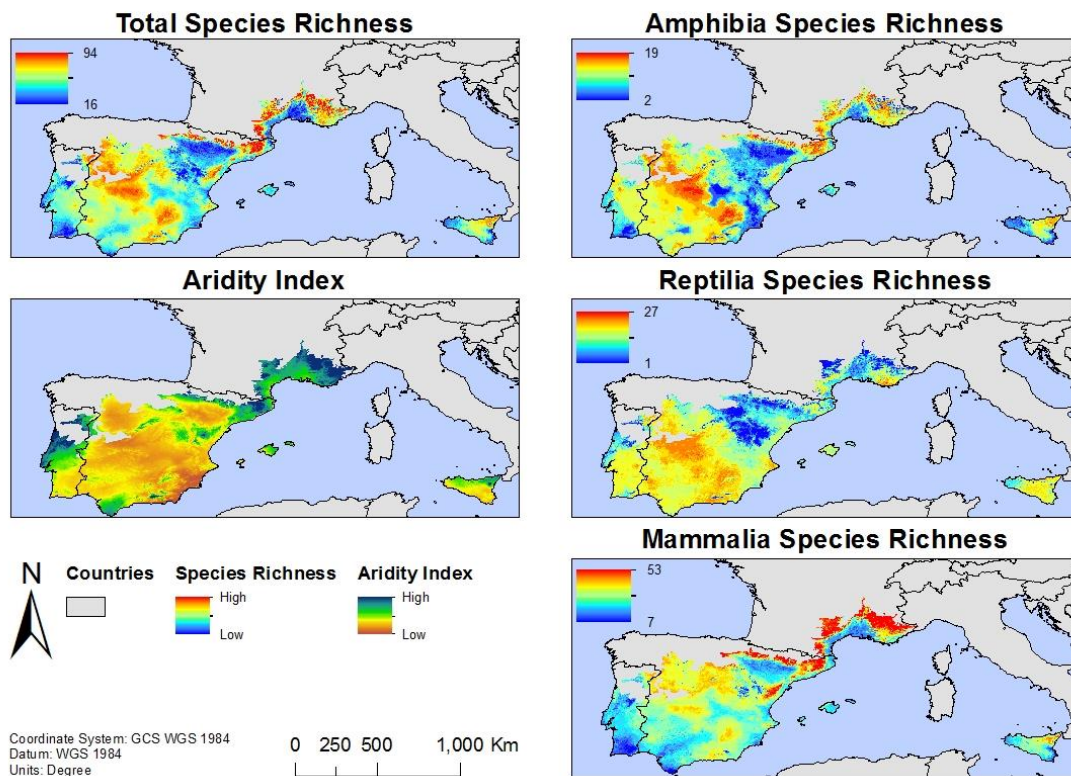


Figure 49: Species richness and aridity index for the Southern Europe dryland test site.

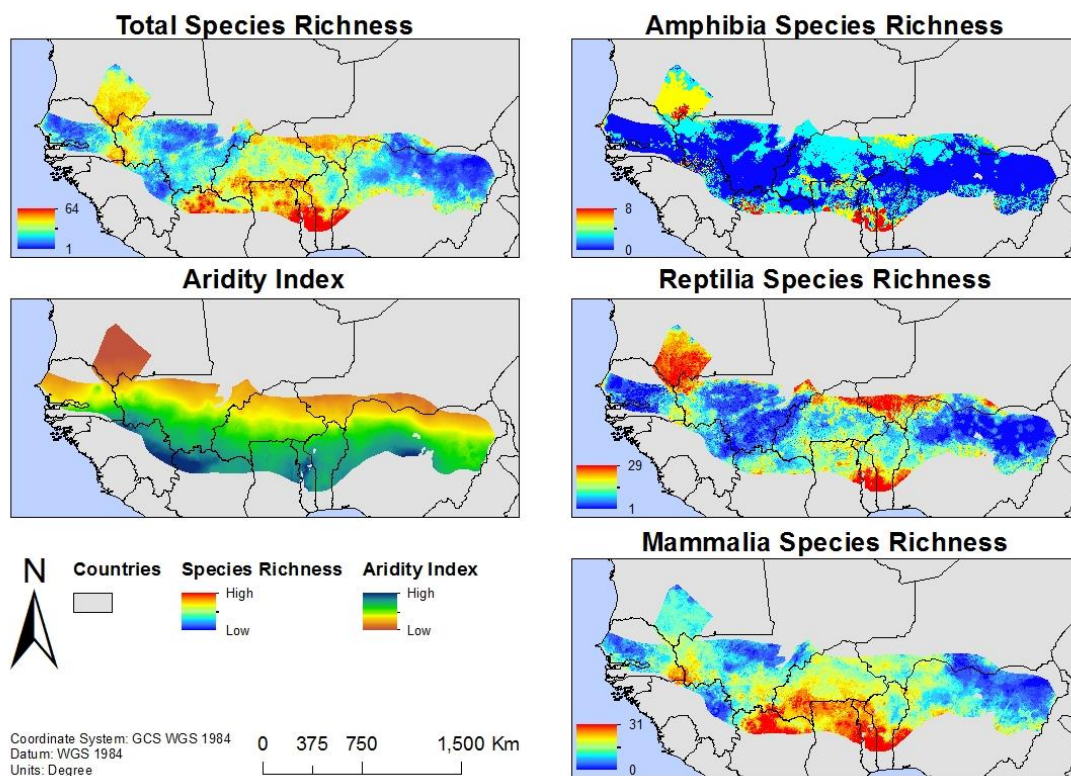


Figure 50: Species richness and aridity index for the West Sudanian dryland test site.

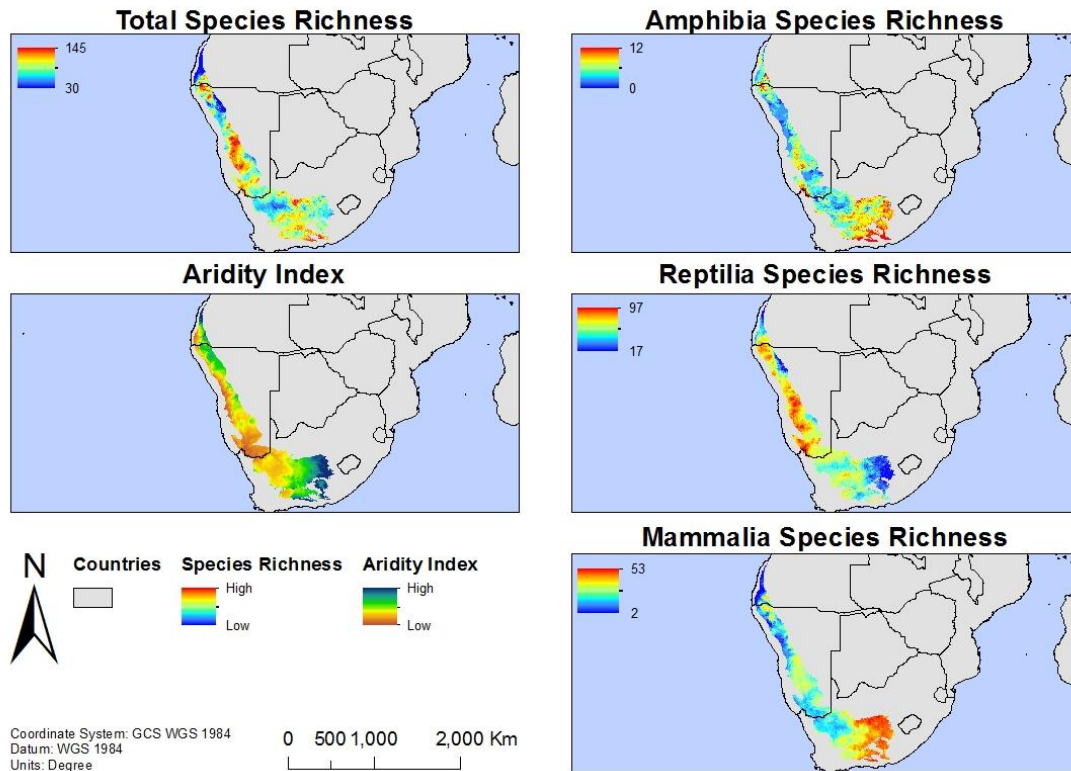


Figure 51: Species richness and aridity index for the South Africa dryland test site.

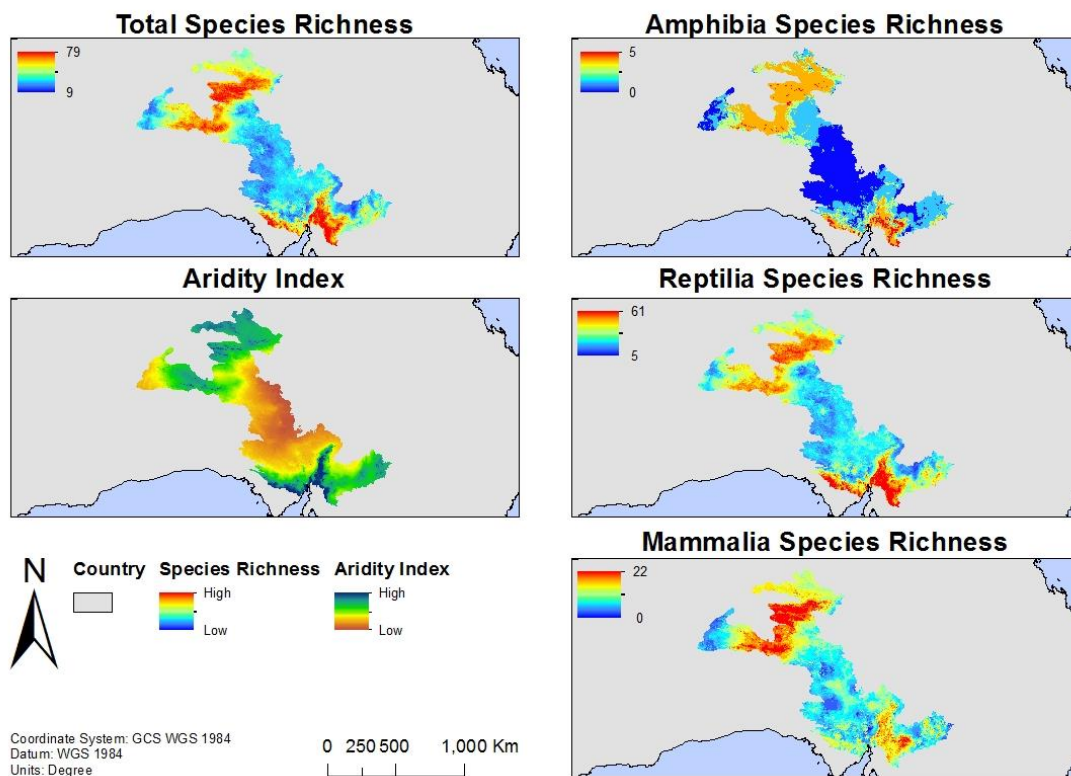


Figure 52: Species richness and aridity index for the Australien dryland test site.

2) Correlation analysis

Correlation analysis between biodiversity indicators and species richness retrieved a low number of statistically significant relationships (above 0.5 absolute value), with most coefficients varying between -0.5 and 0.5. These results hamper strong inferences about general patterns. Nevertheless, it is worth noticing that relationships (positive or negative) were consistent throughout the test sites (Table 12). Within test sites, differences were retrieved between different taxonomic groups, but the majority of those cases had low R^2 scores, to the exception of reptiles in South Africa.

Even though the majority of coefficients are below 0.5 (absolute value), when looking at Figure 53 and Figure 54 we observe a fair match between areas of greater species richness and greater primary productivity as well as longer length of the vegetation season. Both patterns are particularly clear in the South African and Australien test sites, as well as in some parts of the Caatinga.

Low correlation values could be related with non-linear relationships (polynomial, logarithmic, among others), as discussed below in section 3). Other confounding factors could also be responsible for the low correlation values between biodiversity indicators and species richness, namely related with human pressures. Long-term human presence, as observed in South Europe, may induce high degree of habitat destruction/modification through agriculture, urban expansion, modified fire regimes, hunting pressure, and local extirpation of fauna. All of these factors are missing from the individual SDMs, which could lead to the over-prediction of species distribution and subsequently of species richness. Another issue that could have contributed to the low correlation scores is related with pixel resolution. A coarser pixel size probably would have reduced random and local effects associated to species richness variability. Additionally, using distinct pixel sizes for the correlation analyses according to the life-history characteristics of the different taxonomic groups under study may improve results, given the large differences in body size and home ranges of the different species under analyses. The species datasets include from small sized and water-restricted amphibians to large-sized mammals with huge home-ranges, which may affect the ability of SDMs to derive common patterns between species richness and indicators using the same pixel size.

Lastly, the few significant results that were obtained suggest in almost all cases positive relationships between species richness and primary productivity (mean values of each indicator), and negative relationship with variation of those same indicators (only one reported example of a positive correlation between variation of an indicator and biodiversity). Thus, high values of species richness apparently are related with areas of high and stable primary productivity. These results are comparable with macro-scale biogeographical patterns of global distribution of species richness, in which high species richness is mostly found in climatically more stable regions, such as in the tropics (Lomolino et al. 2010).

Table 12: Summary of statistically significant relationships between species richness and biodiversity indicators, for drylands. For each test site, species richness are indicated as - T (Total), A (Amphibia), M (Mammalia) and R (Reptilia). Positive relationships are coloured from light to dark green, while negative are coloured from yellow to red.

			Caatinga				S Europe				W Sudanian				S Africa				Australien			
Indicators			T	A	M	R	T	A	M	R	T	A	M	R	T	A	M	R	T	A	M	R
fAPAR	Cyclic Fraction	Mean																				
		Var																				
	Dry Season	Mean																				
		Var																				
	Vegetation Year	Mean																				
		Var																				
Rainfall	Cyclic Fraction	Mean																				
		Var																				
	Vegetation Year	Mean																				
		Var																				
Rain Use Efficiency	Cyclic Fraction	Mean																				
		Var																				
	Dry Season	Mean																				
		Var																				
	Vegetation Year	Mean																				
		Var																				
Soil Moisture	Cyclic Fraction	Mean																				
		Var																				
	Vegetation Year	Mean																				
		Var																				
Soil Moisture Use	Cyclic Fraction	Mean																				
		Var																				
	Dry Season	Mean																				

		Vegetation Year													
Efficiency	Var														
	Mean														
	Var														
Length Of Vegetation Season	Mean														
	Var														

R ² >	Positive	Negative
0.2		
0.3		
0.4		
0.5		

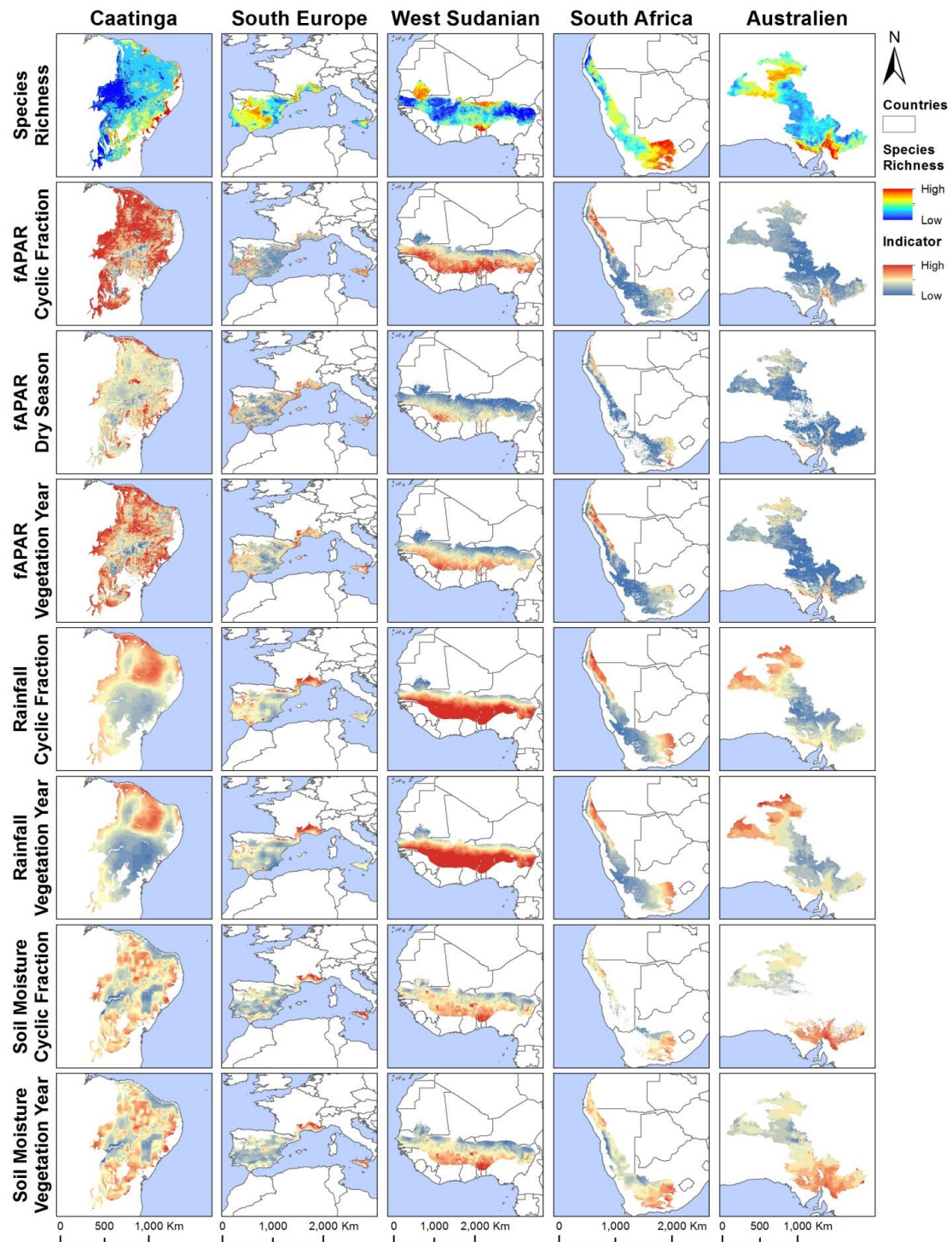


Figure 53: Geographic distribution of mean values of RS indicators and species richness for each dryland test site. From left to right: Caatinga - reptiles; South Europe - amphibians; West Sudanian - reptiles; South African - mammals; Australien - reptiles.

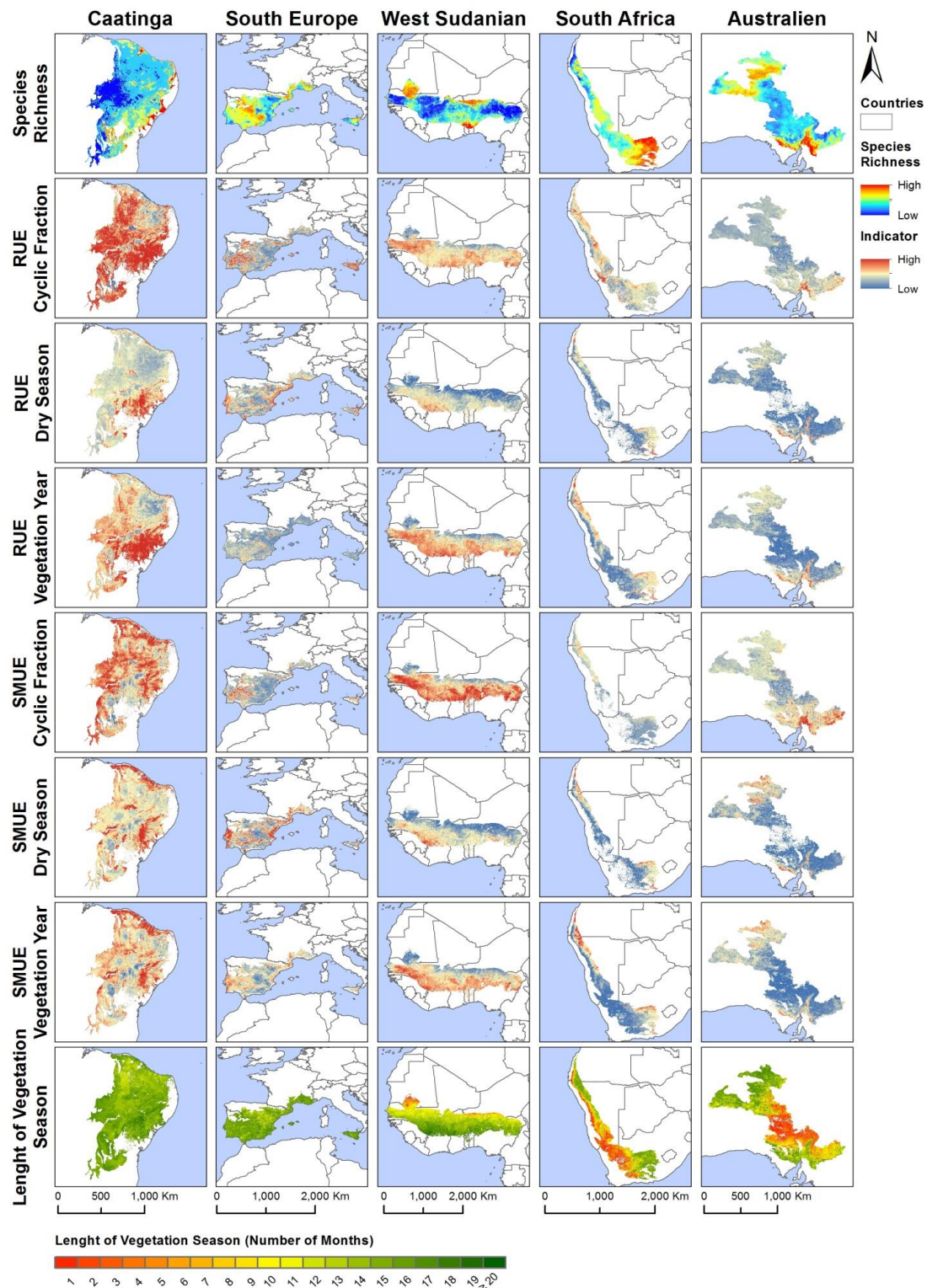


Figure 54: Geographic distribution of mean values of RS indicators and species richness for each dryland test site. From left to right: Caatinga - reptiles; South Europe - amphibians; West Sudanian - reptiles; South African - mammals; Australien - reptiles.

In the Caatinga test site only one significant correlation was retrieved (Table 13), RUE Dry Season Mean, and it is displayed in Figure 55. There is fair match between species diversity and RUE during the dry season. High values can be found for both variables along the south-

eastern limits of the study area and also in some smaller ranges in the centre. Low values are also coincident, especially the large central areas.

Table 13: Correlation coefficient values between biodiversity indicators and species richness (total and for each taxonomic group) for the Caatinga dryland test site.

			Species Richness			
Indicators			Total	Amphibia	Mammalia	Reptilia
fAPAR	Cyclic Fraction	Mean	-0.1125	-0.0886	-0.0975	-0.0873
		Var	0.2124	0.1600	0.1591	0.2243
	Dry Season	Mean	0.4955	0.4292	0.3900	0.4134
		Var	0.3040	0.1570	0.2914	0.2846
	Vegetation Year	Mean	0.2724	0.2343	0.2210	0.2162
		Var	-0.0582	-0.0954	-0.0438	0.0050
Rainfall	Cyclic Fraction	Mean	-0.1619	-0.0089	-0.2280	-0.1200
		Var	0.0553	0.1225	-0.0688	0.1645
	Vegetation Year	Mean	-0.1574	-0.0055	-0.2241	-0.1161
		Var	0.0592	0.1234	-0.0647	0.1694
Rain Use Efficiency	Cyclic Fraction	Mean	-0.0171	-0.1271	0.0556	-0.0102
		Var	0.3246	0.2291	0.2715	0.3052
	Dry Season	Mean	0.5467	0.3600	0.4999	0.4608
		Var	0.3341	0.2124	0.2917	0.3193
	Vegetation Year	Mean	0.3510	0.1723	0.3565	0.2996
		Var	0.1706	0.0975	0.1363	0.2027
Soil Moisture	Cyclic Fraction	Mean	0.1821	0.1516	0.1684	0.1180
		Var	-0.2019	-0.1543	-0.1633	-0.1949
	Vegetation Year	Mean	0.2651	0.2046	0.2505	0.1821
		Var	-0.2655	-0.1840	-0.2355	-0.2399
Soil Moisture Use Efficiency	Cyclic Fraction	Mean	-0.1687	-0.1218	-0.1590	-0.1186
		Var	0.2330	0.1699	0.1997	0.2039
	Dry Season	Mean	0.3486	0.3051	0.2678	0.3060
		Var	0.2265	0.1200	0.2366	0.1653
	Vegetation Year	Mean	0.1322	0.1430	0.0775	0.1322
		Var	-0.0953	-0.1050	-0.0479	-0.1098
Length Of Vegetation Season		Mean	0.3168	0.1976	0.3099	0.2404

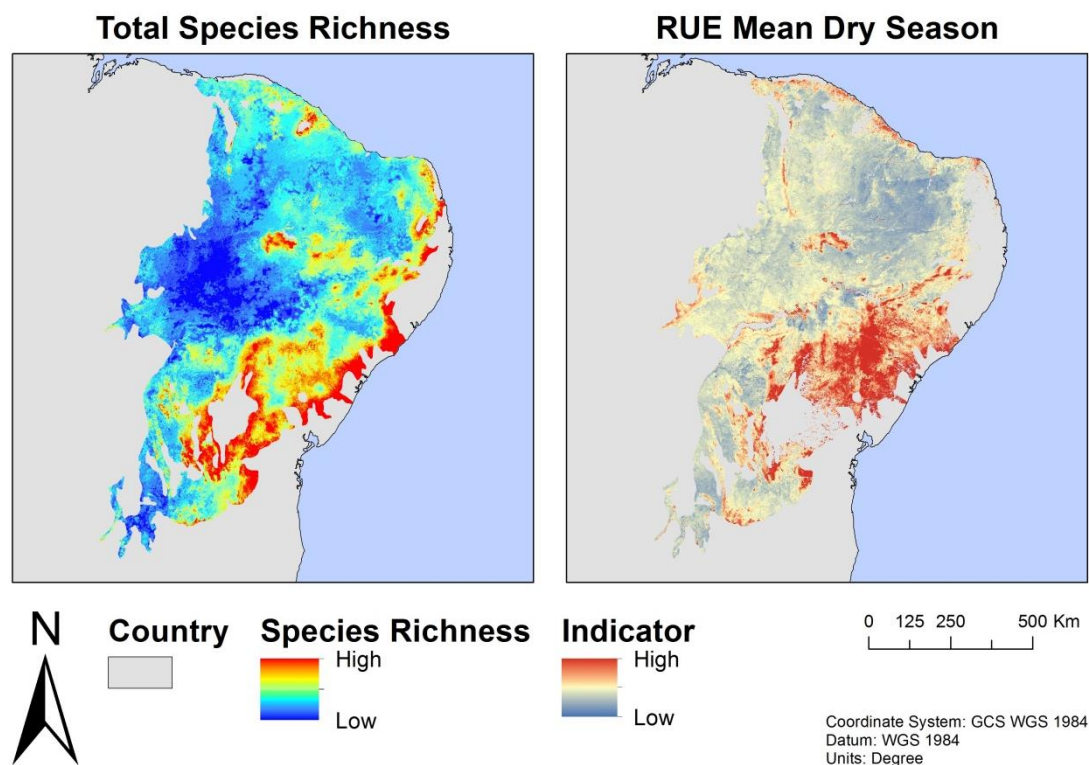


Figure 55: Total species richness and RUE Mean Dry Season for the Caatinga dryland test site.

The South Europe test site did not retrieve any significant correlation (Table 14). In regions where human activities are very intense, such as in South Europe, biodiversity distribution is probably most influenced by agricultural landscapes, resource exploration (forestry activities), fire regimes, and road connectivity, among others. Indicators associated with human activities are worth exploring in the future.

Table 14: Correlation coefficient values between biodiversity indicators and species richness (total and for each taxonomic group) for the South Europe dryland test site.

			Species Richness			
Indicators			Total	Amphibia	Mammalia	Reptilia
fAPAR	Cyclic Fraction	Mean	0.0907	0.1332	0.0418	0.0557
		Var	-0.1005	-0.0977	-0.1665	0.1055
	Dry Season	Mean	0.0844	0.1442	0.1471	-0.1458
		Var	-0.0881	-0.1699	-0.0633	0.0137
	Vegetation Year	Mean	0.1324	0.2082	0.1632	-0.1017
		Var	-0.1237	-0.1239	-0.2104	0.1419
Rainfall	Cyclic Fraction	Mean	0.2477	0.3055	0.3596	-0.2214
		Var	-0.1224	0.0300	-0.3480	0.2663
	Vegetation Year	Mean	0.2414	0.2858	0.4001	-0.2909
		Var	-0.1078	0.0710	-0.4049	0.3688
Rain Use Efficiency	Cyclic Fraction	Mean	-0.0009	0.0329	-0.0943	0.1336
		Var	-0.1229	-0.0728	-0.2487	0.1714

	Dry Season	Mean	-0.0092	0.0356	-0.0096	-0.0320	
		Var	-0.1183	-0.1620	-0.0988	-0.0061	
	Vegetation Year	Mean	-0.1750	-0.0957	-0.3118	0.1669	
		Var	-0.1750	-0.0957	-0.3118	0.1669	
Soil Moisture	Cyclic Fraction	Mean	0.1426	0.0190	0.3282	-0.2114	
		Var	0.0001	0.1024	-0.1540	0.1810	
	Vegetation Year	Mean	0.1410	-0.0036	0.3833	-0.2891	
		Var	0.0585	0.2010	-0.2054	0.3357	
Soil Moisture Use Efficiency	Cyclic Fraction	Mean	0.0923	0.1844	-0.0236	0.1295	
		Var	-0.1291	-0.1509	-0.1660	0.0707	
	Dry Season	Mean	0.0705	0.1524	0.0806	-0.0714	
		Var	-0.1017	-0.1852	-0.0673	-0.0022	
	Vegetation Year	Mean	0.1095	0.2434	0.0398	0.0250	
		Var	-0.1327	-0.1511	-0.1737	0.0744	
	Length Of Vegetation Season		Mean	0.0919	0.0983	-0.0164	0.1825

As for the West Sudanian Savannah, two significant correlations were retrieved (Table 15). Mammalian species richness showed to be related with the SMUE Mean cyclic fraction and vegetation year (Figure 56). High values are found in the southern range of the test site, where water availability is greater (and hence also soil moisture), while the more arid regions in the northern areas of the test site hold less species.

Table 15: Correlation coefficient values between biodiversity indicators and species richness (total and for each taxonomic group) for the West Sudanian Savannah dryland test site.

Indicators			Species Richness			
			Total	Amphibia	Mammalia	Reptilia
fAPAR	Cyclic Fraction	Mean	0.0665	-0.2492	0.3746	-0.2558
		Var	0.0673	0.3007	-0.2154	0.3244
	Dry Season	Mean	0.1474	-0.0916	0.3823	-0.1425
		Var	0.0694	0.2807	-0.2285	0.3583
	Vegetation Year	Mean	0.1134	-0.1892	0.4126	-0.2190
		Var	0.0684	0.3043	-0.2284	0.3412
Rainfall	Cyclic Fraction	Mean	0.1315	-0.1815	0.4567	-0.2398
		Var	-0.1432	0.0626	-0.3139	0.0688
	Vegetation Year	Mean	0.1363	-0.1784	0.4614	-0.2364
		Var	-0.2974	-0.0453	-0.4561	-0.0560
Rain Use Efficiency	Cyclic Fraction	Mean	-0.1041	-0.2808	0.1333	-0.3059
		Var	0.0694	0.2945	-0.2163	0.3307
	Dry Season	Mean	0.0575	-0.1507	0.3026	-0.2167
		Var	0.0352	0.2696	-0.2736	0.3455
	Vegetation Year	Mean	-0.0389	-0.2680	0.2543	-0.3202
		Var	0.0688	0.3065	-0.2413	0.3561

Soil Moisture	Cyclic Fraction	Mean	0.3851	0.1376	0.5464	0.1081
		Var	-0.2009	-0.0082	-0.3090	-0.0406
	Vegetation Year	Mean	0.3503	0.0902	0.5583	0.0318
		Var	-0.2627	-0.0879	-0.2996	-0.1624
Soil Moisture Use Efficiency	Cyclic Fraction	Mean	-0.0952	-0.3178	0.1871	-0.3467
		Var	0.0412	0.2823	-0.2388	0.3069
	Dry Season	Mean	0.0206	-0.1928	0.2753	-0.2501
		Var	0.0832	0.2902	-0.2161	0.3694
	Vegetation Year	Mean	-0.0851	-0.3179	0.2173	-0.3583
		Var	0.0905	0.3175	-0.2011	0.3502
Length Of Vegetation Season		Mean	0.0096	-0.2483	0.3279	-0.3140

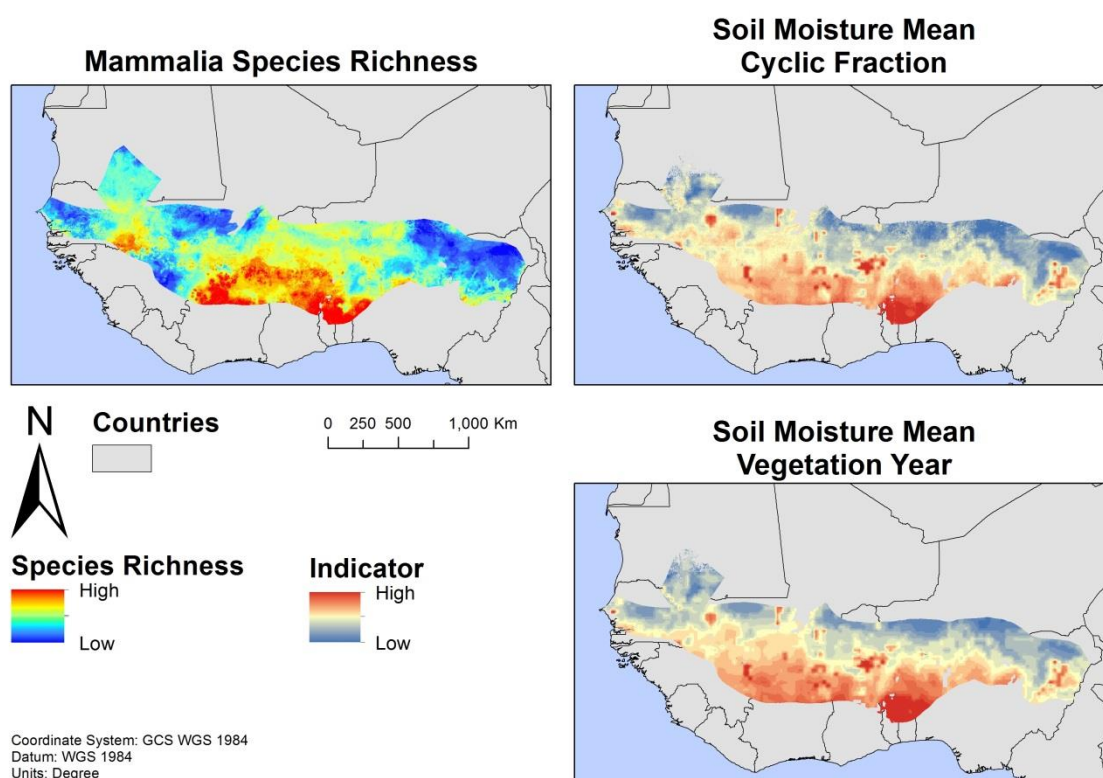


Figure 56: Mammalia species richness, SMUE Mean Cyclic Fraction and SMUE Vegetation Year for the West Sudanian Savannah dryland test site.

Regarding the South African test site, analysis retrieved eight statistically significant correlations (Table 16), of which seven are related with mammalian species richness and the remaining one with reptile diversity (Rainfall variation for the Vegetation Year). Of the seven correlations with mammal diversity, six are negative (variation of fAPAR, Rainfall, RUE and SMUE indicators) and a positive relationship was retrieved with the Length of the Vegetation Season.

The two strongest correlations (one positive and one negative) are displayed in Figure 57. Within the South African test site, mammal diversity seems to be higher in areas where vegetation is more stable through time, in more inland regions further away from the

influence of the Namibian desert. The opposite occurs in the more arid areas where vegetation is more seasonal, and so species richness decreases.

Table 16: Correlation coefficient values between biodiversity indicators and species richness (total and for each taxonomic group) for the South African dryland test site.

			Species Richness			
Indicators			Total	Amphibia	Mammalia	Reptilia
fAPAR	Cyclic Fraction	Mean	0.0156	0.2005	0.2947	-0.2405
		Var	-0.0866	-0.3946	-0.5215	0.3665
	Dry Season	Mean	-0.1155	0.3211	0.3262	-0.4077
		Var	0.1759	-0.2718	-0.3336	0.4725
	Vegetation Year	Mean	0.0156	0.2005	0.2947	-0.2405
		Var	-0.0577	-0.4416	-0.5713	0.4426
Rainfall	Cyclic Fraction	Mean	-0.0474	0.3056	0.4411	-0.4370
		Var	-0.0794	-0.4460	-0.5940	0.4378
	Vegetation Year	Mean	-0.0723	0.3043	0.4485	-0.4701
		Var	0.0236	-0.4849	-0.5637	0.5307
Rain Use Efficiency	Cyclic Fraction	Mean	0.1887	0.1619	0.0678	0.1307
		Var	-0.0735	-0.3744	-0.4866	0.3508
	Dry Season	Mean	-0.0675	0.3303	0.3139	-0.3479
		Var	0.1651	-0.2793	-0.3460	0.4712
	Vegetation Year	Mean	0.0765	0.3526	0.3937	-0.2714
		Var	-0.0287	-0.4297	-0.5514	0.4570
Soil Moisture	Cyclic Fraction	Mean	-0.1669	0.3128	0.2469	-0.3970
		Var	0.0170	-0.4771	-0.3745	0.3456
	Vegetation Year	Mean	-0.0587	0.4702	0.4559	-0.4810
		Var	-0.0273	-0.3290	-0.1943	0.1644
Soil Moisture Use Efficiency	Cyclic Fraction	Mean	-0.0219	-0.1717	-0.1252	0.0876
		Var	0.0194	-0.1152	-0.1556	0.1482
	Dry Season	Mean	-0.0993	0.2736	0.3014	-0.3650
		Var	0.1728	-0.2635	-0.3288	0.4631
	Vegetation Year	Mean	0.0109	0.2597	0.3522	-0.2978
		Var	-0.0596	-0.4234	-0.5577	0.4275
Length Of Vegetation Season		Mean	0.0861	0.4113	0.5494	-0.3898

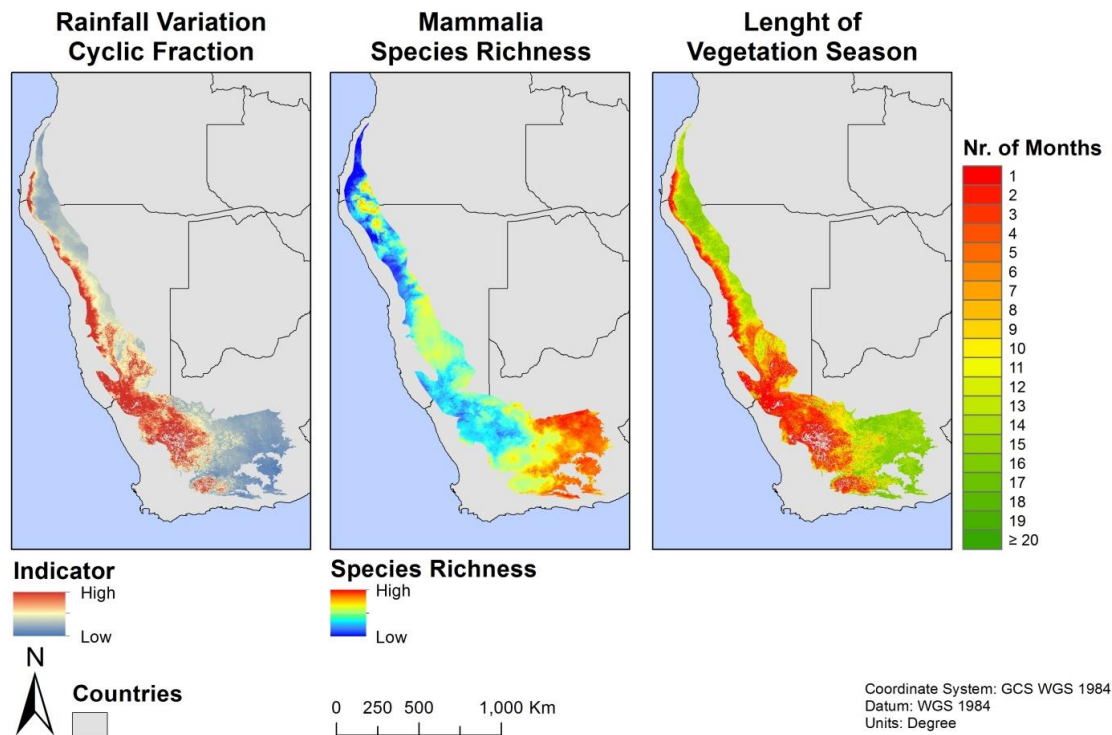


Figure 57: From left to right, Rainfall Variation Cyclic Fraction, Mammalian species richness and Length of Vegetation Season for the South African dryland test site.

Finally, the Australien test site was the one exhibiting the strongest patterns, with 37 significant correlations, some of which with coefficients greater than 0.6 (Table 17). Also, a relationship with all taxonomic groupings was retrieved except with mammals. The majority were positive (25), all related with mean values of indicators, while the remaining 12 were associated with variation of indicators.

Figure 58 illustrates the best examples with Amphibian richness. Both the most northern and southern areas of the test site present the highest values of biodiversity, as well as primary productivity and environmental stability.

Table 17: Correlation coefficient values between biodiversity indicators and species richness (total and for each taxonomic group) for the Australien dryland test site.

			Species Richness			
Indicators			Total	Amphibia	Mammalia	Reptilia
fAPAR	Cyclic Fraction	Mean	0.4916	0.5357	0.2718	0.5173
		Var	-0.4587	-0.4832	-0.2624	-0.4818
	Dry Season	Mean	0.5069	0.5218	0.3403	0.5123
		Var	-0.4380	-0.4471	-0.2547	-0.4576
	Vegetation Year	Mean	0.6045	0.6307	0.3956	0.6173
		Var	-0.5480	-0.5803	-0.3598	-0.5580
Rainfall	Cyclic Fraction	Mean	0.5587	0.6587	0.4549	0.5276
		Var	-0.5407	-0.5859	-0.3586	-0.5476
	Vegetation Year	Mean	0.5313	0.6295	0.4201	0.5059
		Var	-0.2791	-0.2311	-0.0531	-0.3408
Rain Use	Cyclic Fraction	Mean	0.1982	0.1838	0.0215	0.2453

Efficiency		Var	-0.3481	-0.3511	-0.1653	-0.3801	
		Dry Season	Mean	0.4958	0.4939	0.3206	0.5076
			Var	-0.4055	-0.3927	-0.2347	-0.4267
	Vegetation Year	Mean	0.5718	0.5790	0.3557	0.5931	
			Var	-0.5393	-0.5718	-0.3709	-0.5429
Soil Moisture	Cyclic Fraction	Mean	0.1819	0.0098	-0.1144	0.2951	
		Var	0.0137	0.1864	0.2533	-0.1013	
	Vegetation Year	Mean	0.1585	0.0029	-0.0567	0.2430	
			Var	0.1458	0.2876	0.2995	0.0556
Soil Moisture Use Efficiency	Cyclic Fraction	Mean	0.1485	0.2481	0.0602	0.1524	
		Var	-0.1956	-0.1865	-0.0369	-0.2344	
	Dry Season	Mean	0.4794	0.5333	0.3657	0.4636	
		Var	-0.4602	-0.4805	-0.2932	-0.4704	
	Vegetation Year	Mean	0.5649	0.6342	0.4220	0.5522	
		Var	-0.5675	-0.6145	-0.3999	-0.5666	
Length Of Vegetation Season		Mean	0.5445	0.6201	0.4157	0.5270	

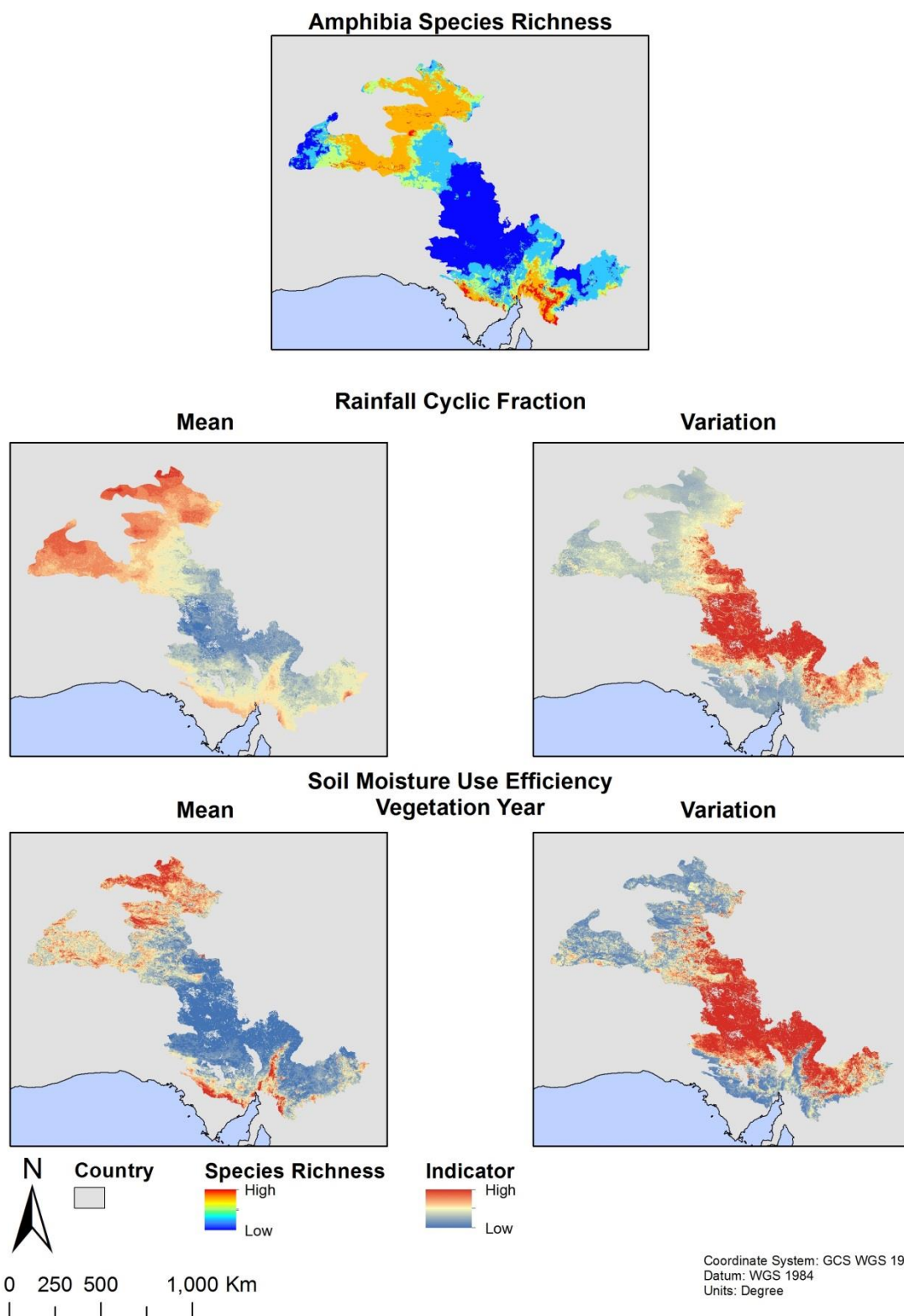


Figure 58: Amphibia species richness, Rainfall Cyclic Fraction (Mean and variation) and SMUE Vegetation Year (Mean and variation) for the Australien dryland test site.

3) *Non-linear relationships*

As for other possible explanatory models, scatter plots of the 1000 random points for each test site are displayed in Figure 59, Figure 60, Figure 61, Figure 62 and Figure 63.

In general, relationships (positive or negative) between biodiversity indicators and species richness followed the same patterns described above in section 2). There are cases in which the relationships are clearly linear, such as fAPAR dry season mean for reptiles in South Africa (plot R3 from Figure 62) and Rainfall cyclic fraction variance for reptiles in Australia (plot R8 from Figure 63). However, there are other examples in which non-linear relationships apparently are better suited, such as with the mammals in West Sudannian Savannah (Figure 61) that follows a logarithmic relation and the RUE dry season mean for mammals in South Africa (plot M13 from Figure 62) that follows an asymptotic relation.

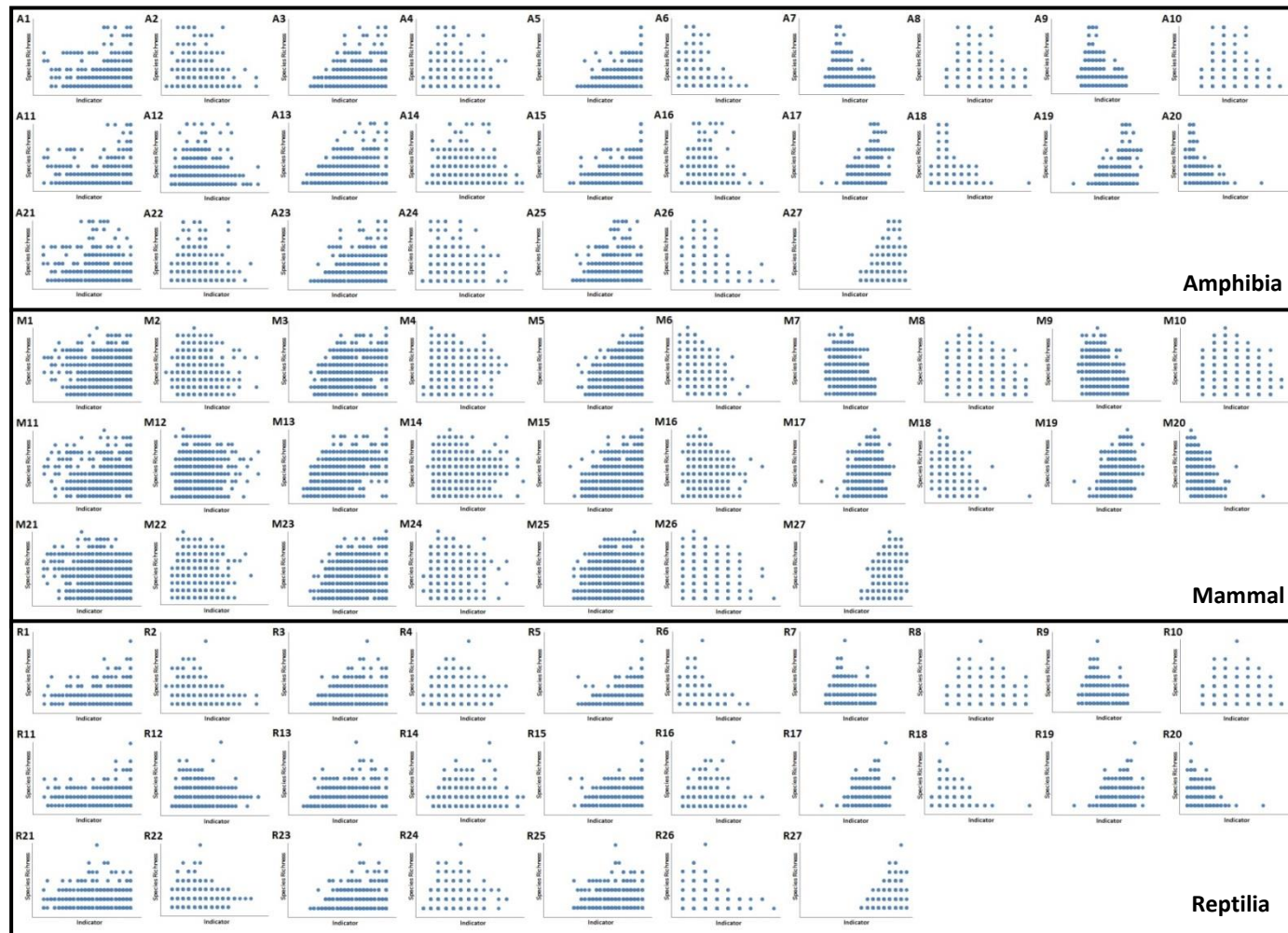


Figure 59: Scatter plots for the Caatinga test site, depicting species richness (Y axis) and RS indicators (X axis) for 1000 points. From top to bottom – Amphibia species richness (A1 to A27), Mammalia species richness (M1 to M27) and Reptilia species richness (R1 to R27). Plots are ordered as in Table 12 – from 1 to 6 (FAPAR); from 7 to 10 (Rainfall); from 11 to 16 (RUE); from 17 to 20 (Soil Moisture); from 21 to 26 (SMUE) and 27 (Length of Vegetation Year).

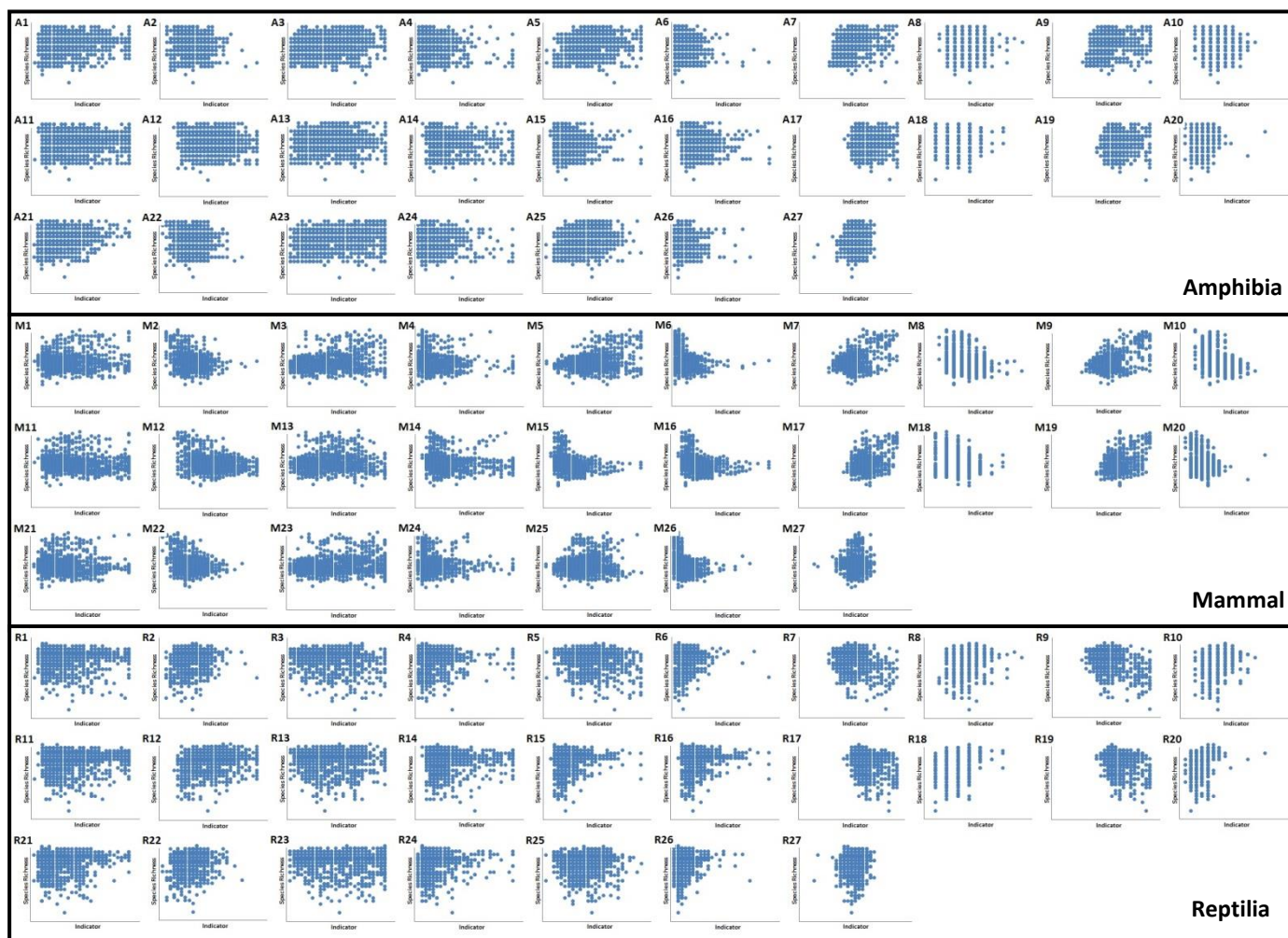


Figure 60: Scatter plots for the South Europe test site, depicting species richness (Y axis) and RS indicators (X axis) for 1000 points. From top to bottom – Amphibia species richness (A1 to A27), Mammalia species richness (M1 to M27) and Reptilia species richness (R1 to R27). Plots are ordered as in Table 12 – from 1 to 6 (fAPAR); from 7 to 10 (Rainfall); from 11 to 16 (RUE); from 17 to 20 (Soil Moisture); from 21 to 26 (SMUE) and 27 (Length of Vegetation Year).

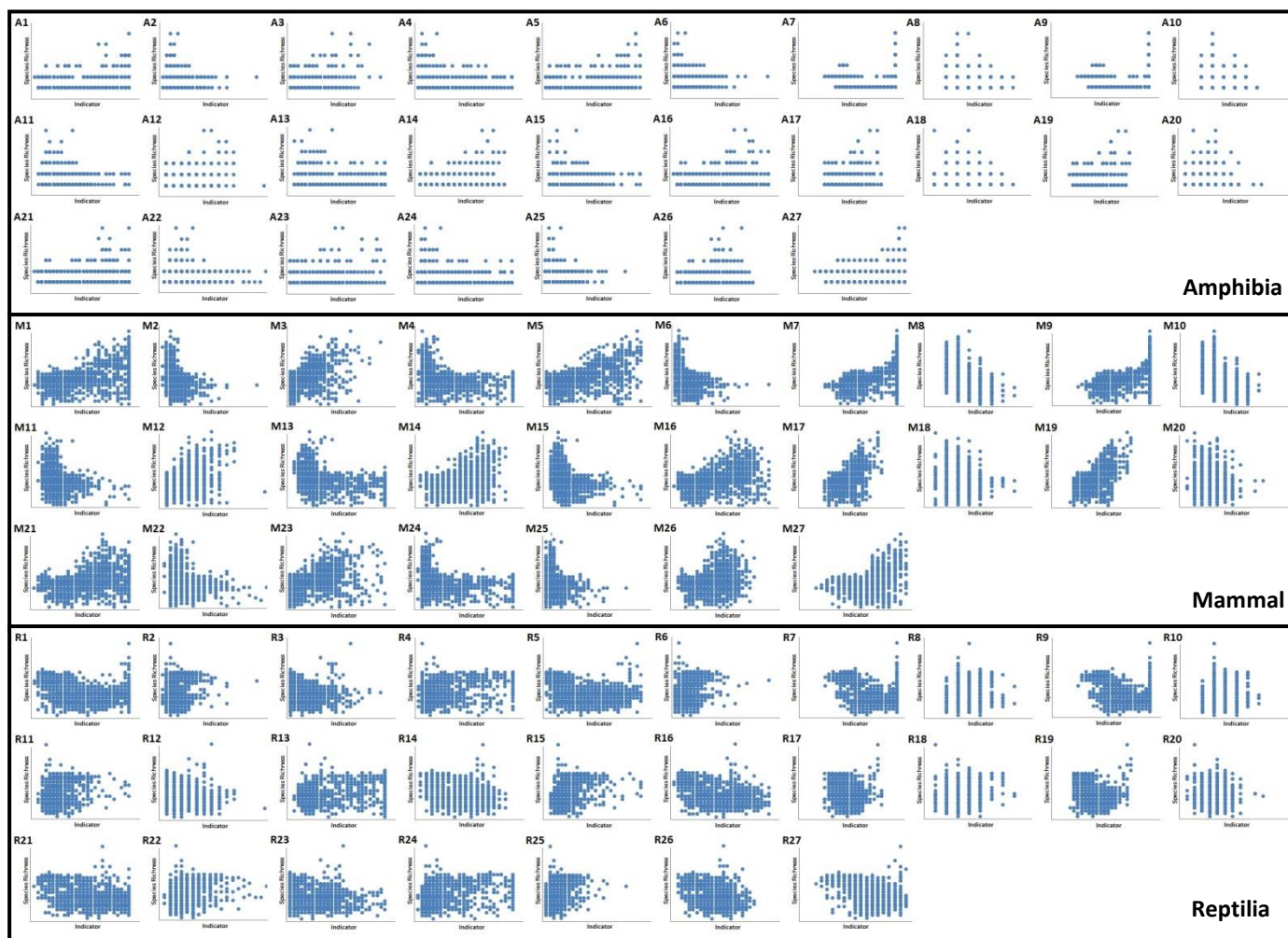


Figure 61: Scatter plots for the West Sudanian Savannah test site, depicting species richness (Y axis) and RS indicators (X axis) for 1000 points. From top to bottom – Amphibia species richness (A1 to A27), Mammalia species richness (M1 to M27) and Reptilia species richness (R1 to R27). Plots are ordered as in Table 12 – from 1 to 6 (fAPAR); from 7 to 10 (Rainfall); from 11 to 16 (RUE); from 17 to 20 (Soil Moisture); from 21 to 26 (SMUE) and 27 (Length of Vegetation Year).

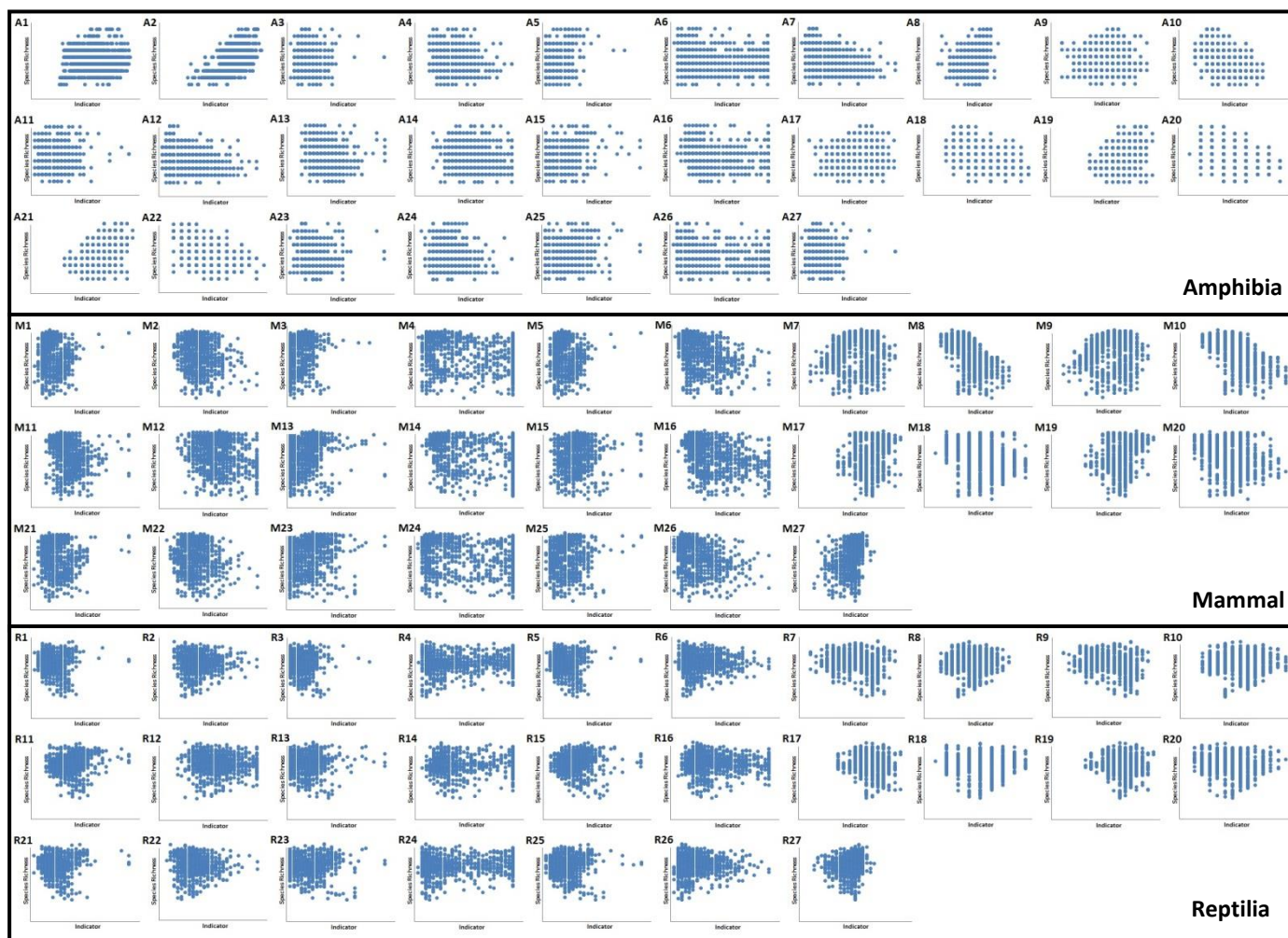


Figure 62: Scatter plots for the South African test site, depicting species richness (Y axis) and RS indicators (X axis) for 1000 points. From top to bottom – Amphibia species richness (A1 to A27), Mammalia species richness (M1 to M27) and Reptilia species richness (R1 to R27). Plots are ordered as in Table 12 – from 1 to 6 (FAPAR); from 7 to 10 (Rainfall); from 11 to 16 (RUE); from 17 to 20 (Soil Moisture); from 21 to 26 (SMUE) and 27 (Length of Vegetation Year).

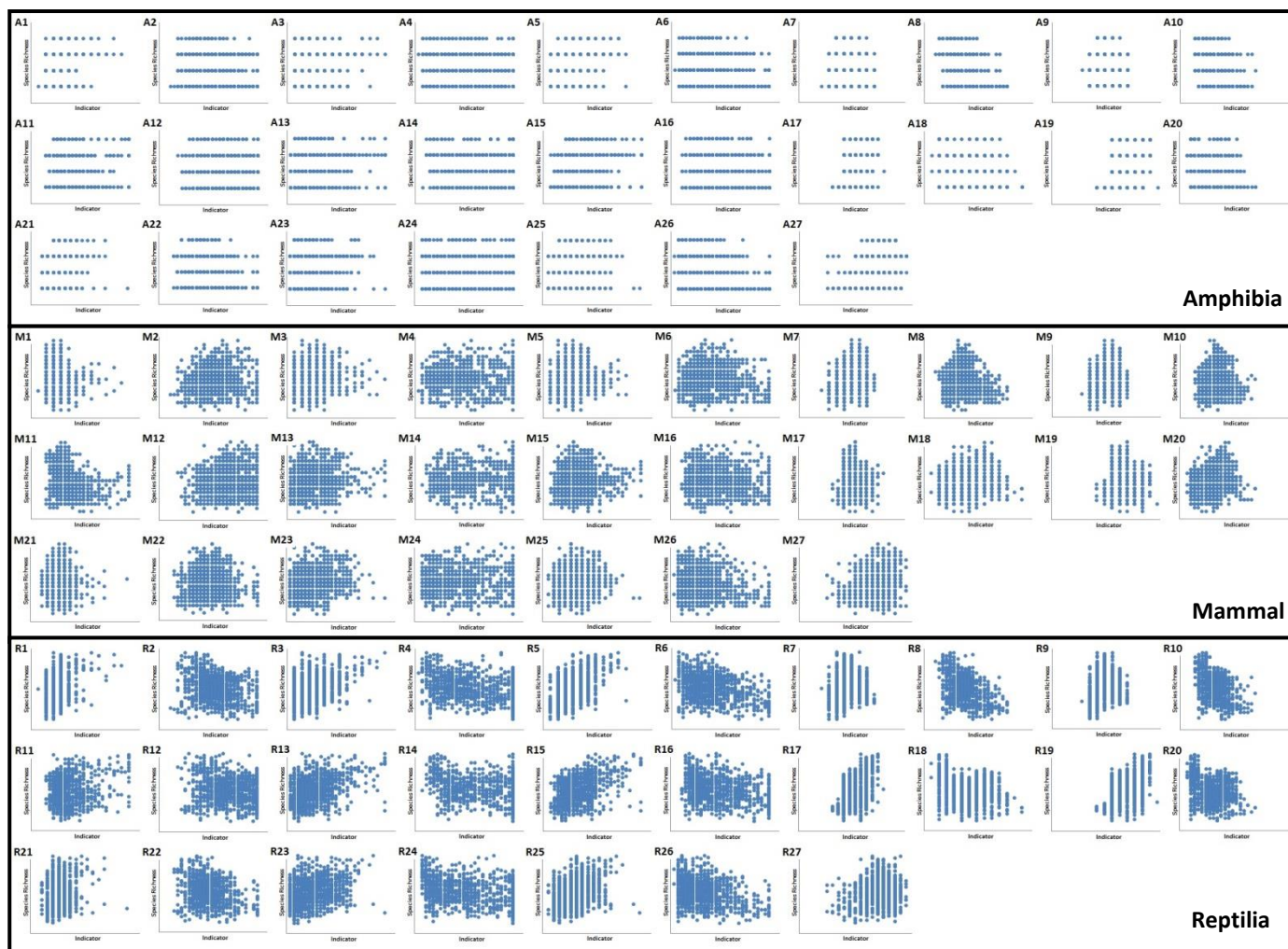


Figure 63: Scatter plots for the Australian test site, depicting species richness (Y axis) and RS indicators (X axis) for 1000 points. From top to bottom – Amphibia species richness (A1 to A27), Mammalia species richness (M1 to M27) and Reptilia species richness (R1 to R27). Plots are ordered as in Table 12 – from 1 to 6 (fAPAR); from 7 to 10 (Rainfall); from 11 to 16 (RUE); from 17 to 20 (Soil Moisture); from 21 to 26 (SMUE) and 27 (Length of Vegetation Year).

3.2.2 Trend

3.2.2.1 Southern Europe

For the Southern-Europe dryland test site, biodiversity trend data was kindly provided by Mário Ferreira and Dr. Pedro Beja, from CIBIO/UP. This data was collected in the coastal plateau of southwest Portugal, within the “Sudoeste Alentejano e Costa Vicentina” National Park (Figure 64). The dataset comprised information on presence and abundance of amphibians in temporary ponds, measured as the average number of larvae by 30 second dip-netting. There were a total of 50 sampling locations, data being collected in 2002, 2006 and between 2009 and 2011, with a minimum of one and a maximum of four sampling dates per year, corresponding in general to the months of February, March, April and June (Ferreira & Beja, 2013).

The dataset comprised abundance values for 13 amphibian taxa, from which we selected the nine most common taxa (presence data and abundance estimates of remaining four taxa were very residual). We calculated average abundances of selected taxa present at least once in time for each pond (from now on just termed abundance). We also counted the number of taxa detected (N Taxa) for each available date of each pond, but this time considering the 13 possible taxa.

Next, we had to assure the maximum overlap in terms of time frame between biodiversity trend data and indicators. The sampling effort was not equal for all ponds, so from the initial 50 we selected only the eight ponds for which we had the best match with indicators, as well as a more complete sampling effort. Still, of the selected eight, five ponds had a high percentage of zero abundance values (either related to the fact that the pond was dry at the time of sampling or no amphibian was detected at all despite the presence of water). Such a high percentage of null values would hamper analysis, so the five mentioned ponds were discarded, leaving a final dataset of three ponds. These are named from now on as test ponds and their locations are depicted in Figure 65.

In addition, while amphibian data in test ponds was organized by days, indicator (fAPAR) values were averaged by fortnight (15 days). As such, the correlations between biodiversity and indicators were essayed using several aggregation schemes as illustrated in Figure 66. We essayed correlations using fAPAR values that corresponded to the fortnight that included the sampling date (Figure 66 – Fortnight 2), the average of the two previous weeks (Figure 66 - Fortnight 1), and the two fortnights after the sampling date (Figure 66 - Fortnights 3 and 4). Additionally, we calculated the monthly fAPAR average between each successive pair

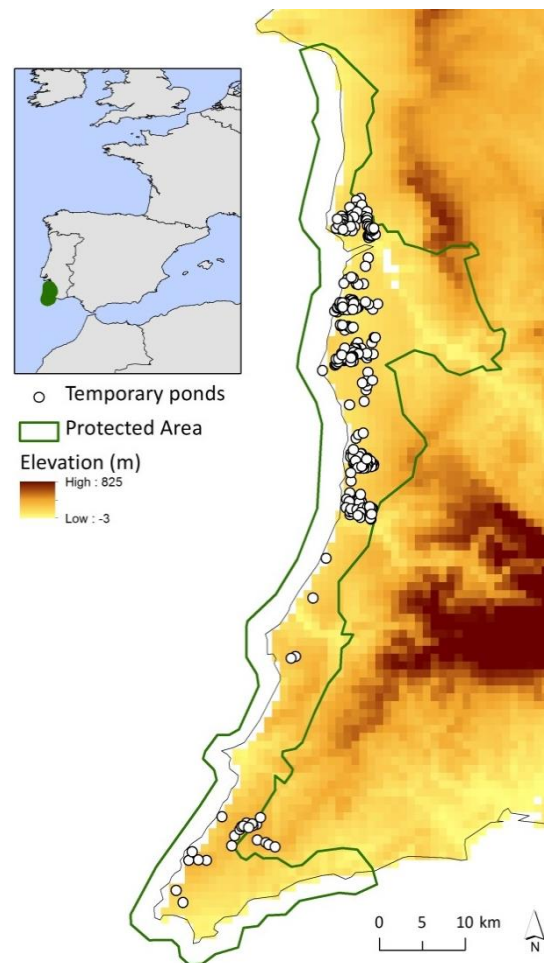


Figure 64: Geographic distribution of the 50 temporary ponds with available amphibian data for South-Europe biodiversity trend data analysis.

of fortnights (Figure 66 – Months 1, 2 and 3), as well as selected the maximum value amongst all fortnights for each sampling date (Maximum, not depicted in Figure 66).

Spearman's rank order correlation analyses were used to assess significance of relationships between biodiversity and indicators, following the same procedure described in chapter 2.2.2.3 for Lake Michigan.

Correlation analysis of trend data for Southern Europe retrieved positive relationships between the biodiversity indicator fAPAR (taken from various time frames) and amphibian abundance in ponds (Table 18). No relevant differences are discernible between the various time frames of fAPAR values (Figure 67 and Figure 68). Of the selected test ponds, A10 showed statistically significant correlations with all the fAPAR estimates and taxa richness (N Taxa) and number of larvae (Abundance). Analysis of test pond G03 did not retrieve any significant correlations with biodiversity data, while test pond C30 demonstrated seven significant relationships, four for the number of taxa and three with abundance. Differences amongst the three ponds might be related with the fact that only pond A10 had permanent water while the other two were temporary. Alternatively, such differences might also be related with the geographic distribution of test ponds (Figure 65), since southern Portugal is in general more arid and with less water available (Portuguese environmental atlas, <http://sniamb.apambiente.pt/webatlas>).

The fAPAR is a measure of primary productivity (proxy for NPP) and it has been proved that higher species diversity is normally related with higher values of primary productivity. These relationships were verified for different taxa, such as tree species (Kale & Roy 2012), invertebrates (Chase & Leibold 2002), birds and mammals (Luck 2007), at both local and regional scales. Plant growth, and hence primary productivity, is very dependent in the availability of water in the form of rain, especially in dry regions such as southern Portugal.

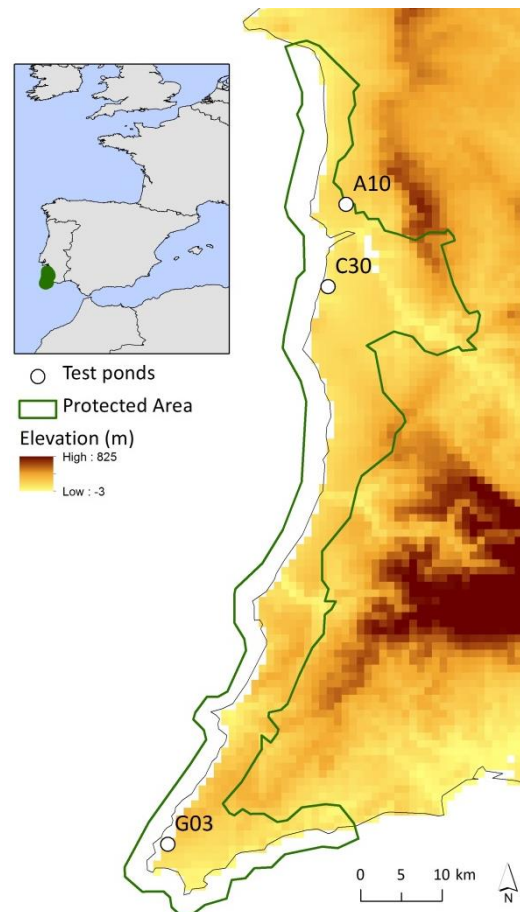


Figure 65: Geographic distribution of the three selected test ponds

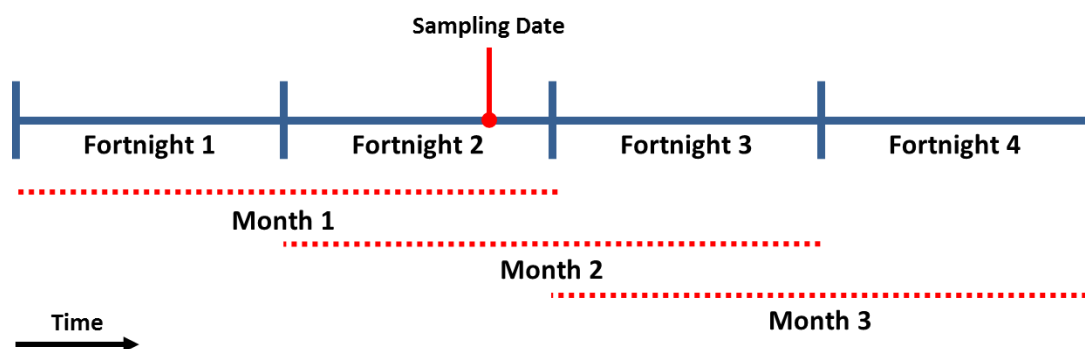


Figure 66: Aggregation schemes of fAPAR values to correlate with biodiversity data.

Amphibians are also very dependent of moisture and often experience rapid and large population expansions after periods of rainfall. So fAPAR could be a viable index of the state of biodiversity, even for water-dependent taxa, such as amphibians.

Table 18: Correlation analysis for Southern Europe trend data. Correlation values are based on Spearman's r , with values above zero indicating a positive correlation, while below zero indicate a negative correlation. Statistically significant probabilities ($p < 0.05$) are depicted between brackets.

Indicators (fAPAR)	N Taxa			Abundance		
	A10	C30	G03	A10	C30	G03
Fortnight 1	0.68 (0.022)	-0.09	0.48	0.74 (0.009)	-0.51	0.43
Fortnight 2	0.58	0.28	0.59	0.67 (0.02)	-0.16	0.53
Fortnight 3	0.66 (0.03)	0.39	0.66 (0.030)	0.76 (0.01)	-0.23	0.68 (0.022)
Fortnight 4	0.66 (0.026)	0.38	0.61 (0.050)	0.83 (0.001)	-0.13	0.82 (0.002)
Month 1	0.62 (0.041)	0.06	0.59	0.69 (0.02)	-0.39	0.53
Month 2	0.66 (0.03)	0.35	0.53	0.75 (0.01)	-0.23	0.52
Month 3	0.66 (0.026)	0.43	0.65 (0.030)	0.83 (0.001)	-0.12	0.73 (0.011)
Maximum	0.66 (0.026)	0.38	0.65 (0.030)	0.72 (0.012)	-0.25	0.57

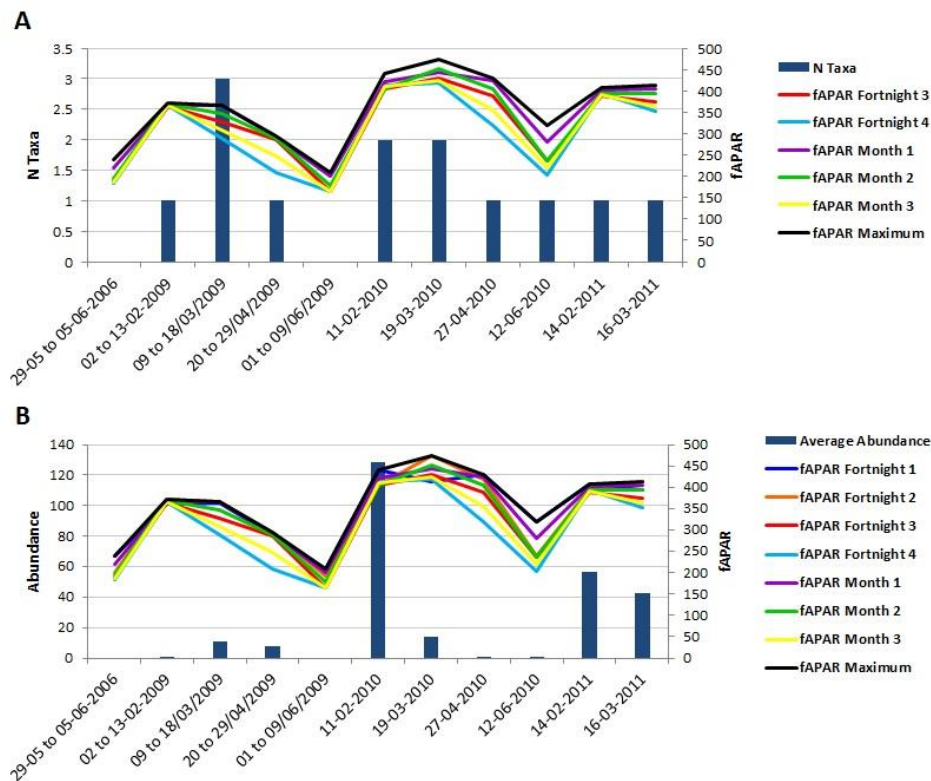


Figure 67: Correlation analysis for test pond A10 based on Spearman's r . Charts depict biodiversity measure (left Y axis) and indicator (right Y axis), per sampling date (X axis): A – Number of amphibian taxa (N Taxa) and fAPAR values; B – Number of amphibian larvae (Abundance) and fAPAR values.

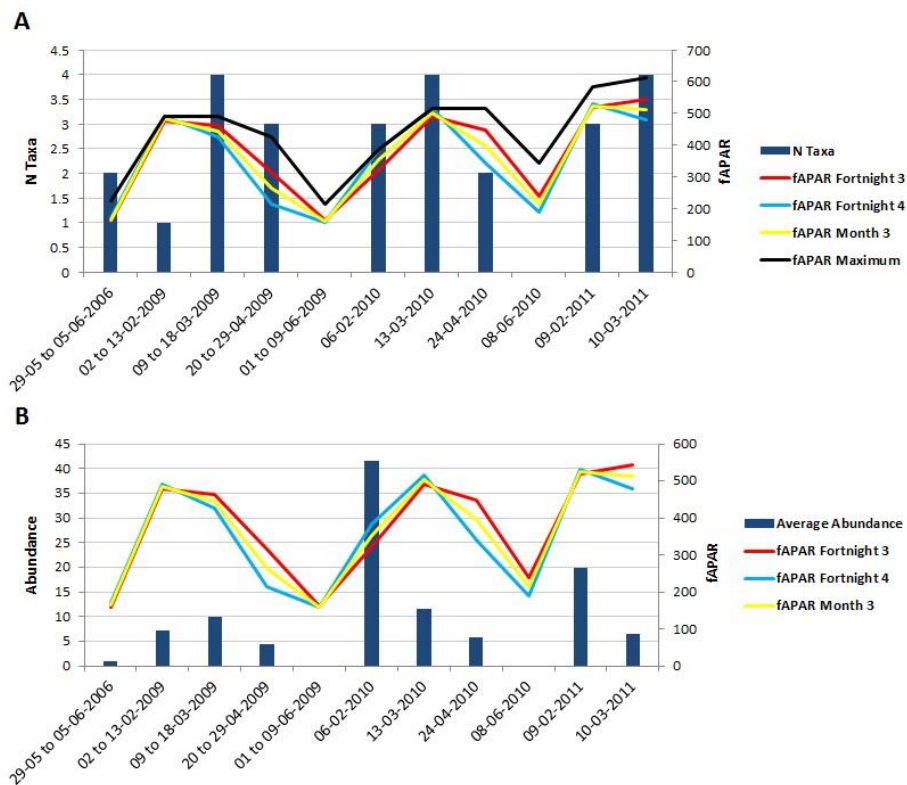


Figure 68: Correlation analysis for test pond G03 based on Spearman's r . Charts depict biodiversity measure (left Y axis) and indicator (right Y axis), per sampling date (X axis): A – Number of amphibian taxa (N Taxa) and fAPAR values; B – Number of amphibian larvae (Abundance) and fAPAR values.

3.2.2.2 South Africa

For the South-Africa dryland test site, biodiversity trend data was kindly provided by Dr. Rita Covas, from CIBIO/UP, and comprises various parameters on the life cycle of the sociable weaver (*Philetairus socius*). The species is known to live in colonies and in general the reproductive season occurs between September and February, in which two breeding events can occur, a first beginning in September/October and a second one starting December/January, since it takes more or less one month to a chick to develop (from the time the egg is laid to when it finally leaves the nest) (Covas et al. 2008). The data was obtained at the Benfontein Game Farm in the Northern Cape Province, South Africa, during 2010 and 2012. The dataset comprised information for various bird colonies on number of eggs (clutch size), number of nests used (as a proxy to the number of breeding birds within a colony) and breeding season length (calculated as the difference between the 10th and the 90th percentile of breeding days starting from the 29th of August). Sampling date was available for data collected per colony between 2010 and 2012, but since the RS indicator fAPAR was averaged by fortnight, clutch size data for example was sum also by fortnight. We selected the eight colonies for which there was the maximum amount of data as possible (Figure 69).

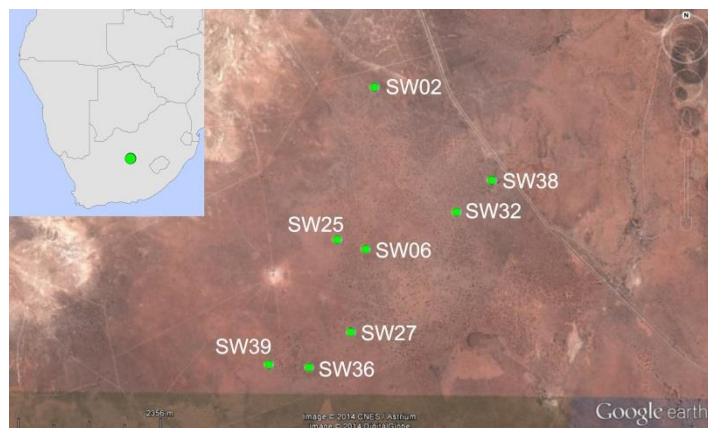


Figure 69: Geographic location of selected sociable weaver colonies.

The first approach consisted of plotting fAPAR values obtained for each of the selected colonies, and visually evaluating if there was a match with the theoretical reproductive season, that lasts between September and February (Figure 70). In general, the reproductive season begins at a low point of fAPAR (low primary productivity), but just closely before a substantial increase and a peak of productivity during December and January. Afterwards, the breeding season ends shortly before a decrease of fAPAR.

Focusing on the time period between 2010 and 2012 (Figure 71 and Figure 72), clutch sizes are larger and egg laying events are more frequent during the second fortnight of December and the first fortnight of February, which matches the highest peak of fAPAR. Also, comparing between the 2010 and 2011 reproductive seasons, it is very clear that the number of eggs, the frequency at which they were laid and the number of nests used were much higher in 2010 than in 2011. In terms of fAPAR, the increased peak of productivity observed in January-February of 2011 is not noticeable during the same time period in 2012. Hence it seems that the primary productivity of the surrounding environment has a positive relationship with breeding success of social weavers. Our results follow the same patterns found by Covas et al. (2008) based on rainfall data.

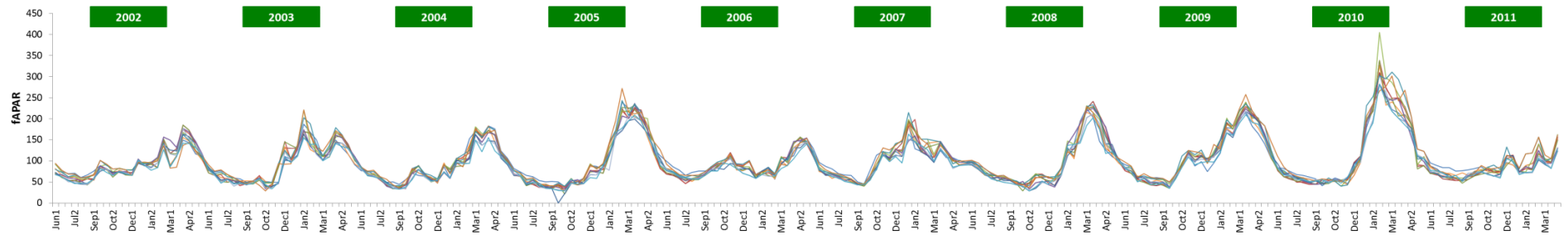


Figure 70: South African trend analyses - plot of fortnight fAPAR values for the eight selected colonies (Y-axis, coloured lines), along with theoretical reproductive season of sociable weavers (green horizontal bars), between June 2002 and April 2012. The year corresponds to the reproductive season and not calendar year.

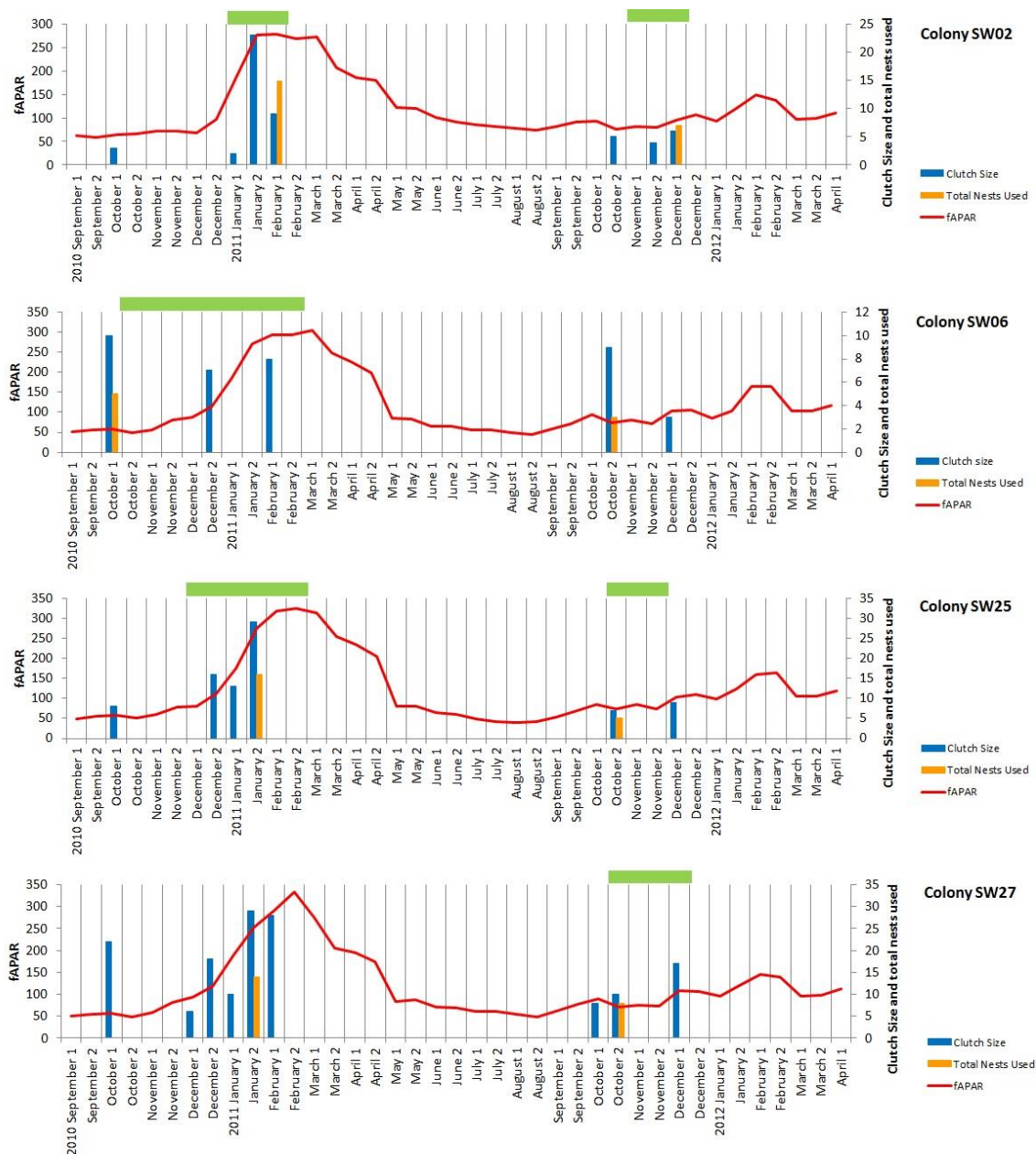


Figure 71: Fortnight fAPAR values (left Y-axis, red line) along with clutch size (right Y-axis, blue vertical bars), total nests used (right Y-axis, orange vertical bars) and breeding season length (green horizontal bars), for the first four colonies out of the eight selected. Time period from September 2010 to April 2012.

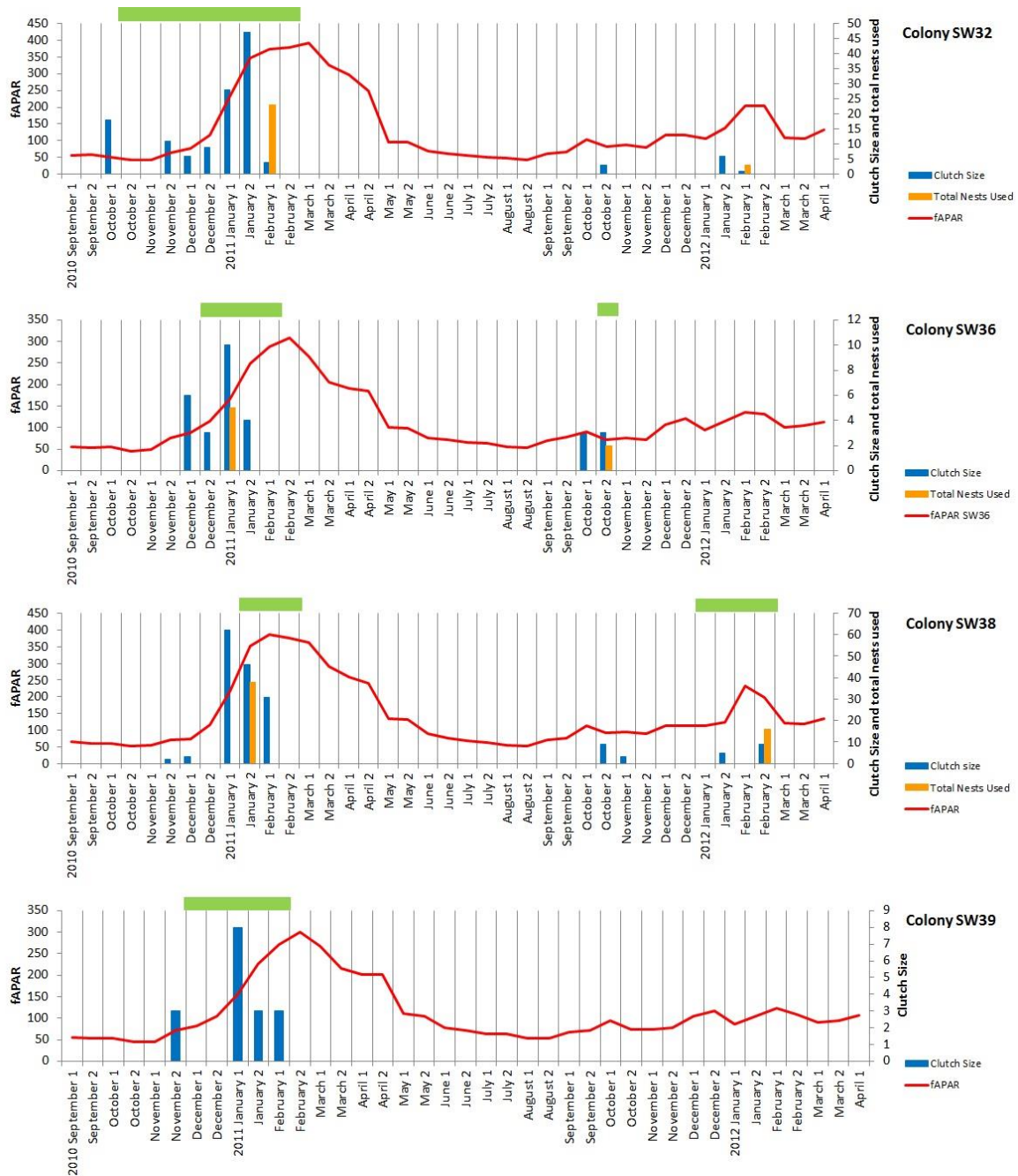


Figure 72: Fortnight fAPAR values (left Y-axis, red line) along with clutch size (right Y-axis, blue vertical bars), total nests used (right Y-axis, orange vertical bars) and breeding season length (green horizontal bars), for the second four colonies out of the eight selected. Colony SW39 did not had data on total nests used. Time period from September 2010 to April 2012.

3.2.2.3 Australien

Biodiversity trend data for the Australien test site was extracted from Pavey & Nano (2013), and consisted on abundance of rodents (capture per 100 trap nights) and raptors (mean encounter rate per km per transect) from October 2007 to May 2011 (11 monthly sampling sessions). The study area had approximately 7000 km² and it was located at the Andado Station in the northwest Simpson Desert, southeast of Alice Springs, Australia (Figure 73).

Due to the uneven sampling effort, strong statistical analyses were not performed. But looking at Figure 74, it is clearly visible that the increase in abundance of both rodents and raptors was coincidental with an increase in fAPAR (primary productivity). The same was reported in Pavey & Nano (2013) relative to an increase in rainfall.

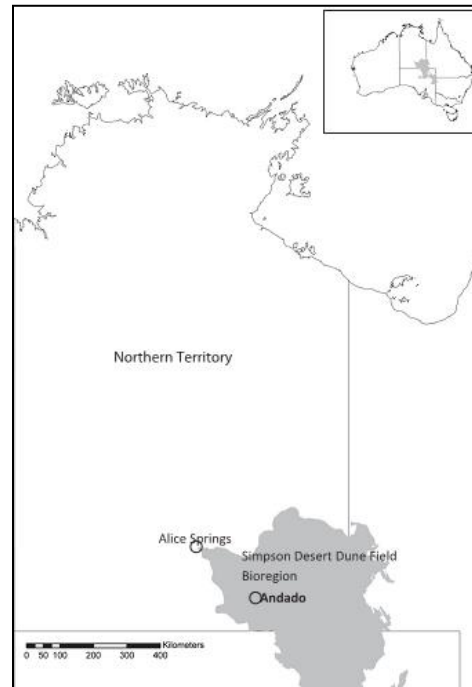


Figure 73: Geographic location of the Andado Station, for the Australien trend data analysis. Adapted from Nano & Pavey (2013).

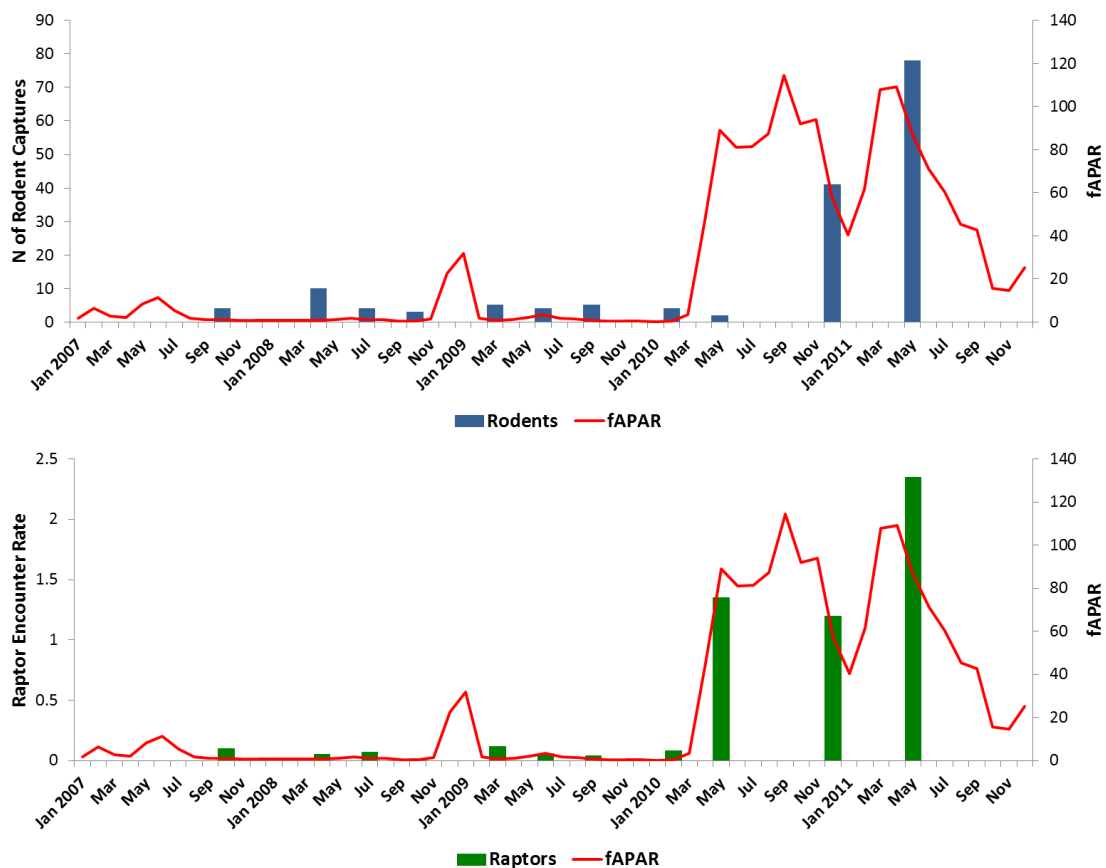


Figure 74: Plot of Australien trend analysis. Charts depict biodiversity measure (left Y axis) and fAPAR (right Y axis), per sampling date (X axis, from January 2007 to December 2011). Horizontal bars represent biodiversity data (blue – rodents; green – raptors), and red line fAPAR.

3.2.2.4 Caatinga

Two sources of biodiversity trend data were available for the Caatinga test site. The first consisted of tadpole abundance and species richness between September 2011 and August 2012, within the “Unidade de Conservação Estadual Monumento Natural Grota do Angico”, state of Sergipe, Brazil (Figure 75) (Silva 2013).



Figure 75: Geographic location of the study area for the first Caatinga trend data analysis. Adapted from Silva (2013).

Biodiversity data was compiled from a series of sampling efforts performed at different ponds, since effort by sampled pond was not continuous throughout time. Thus, a rigorous statistical analysis was not possible and instead only a visual comparison was conducted. Also, since biodiversity data was gathered monthly (a precise date was not provided), fAPAR data was averaged between the second fortnight of one month and the first fortnight of the month after, to encompass any delayed responses to changes in levels of productivity.

From Figure 76, no particular pattern between biodiversity data and productivity can be observed.

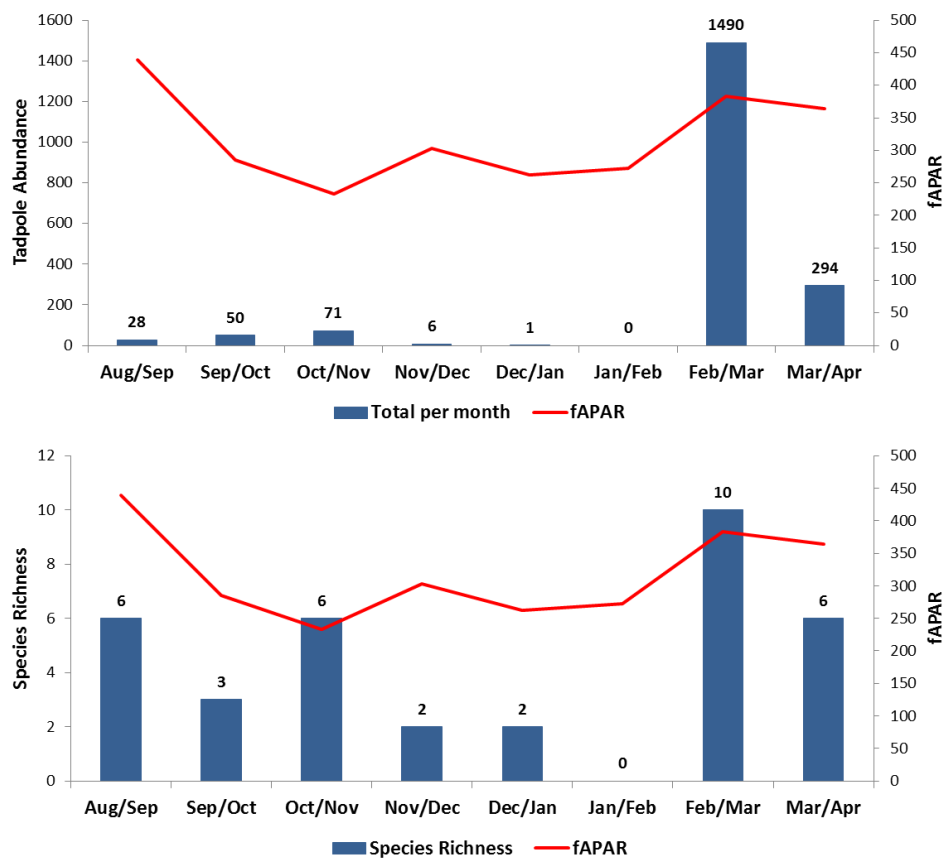


Figure 76: Plot of the first Caatinga trend analysis. Charts depict biodiversity measure (left Y axis) and fAPAR (right Y axis), per sampling date (X axis, from September 2011 to March 2012). Blue horizontal bars represent biodiversity data and red line fAPAR.

The second biodiversity data source consisted of a report (available at <http://dspace.bc.uepb.edu.br:8080/xmlui/handle/123456789/181>) on bird species richness at the “Complexo Aluizio Campos”, Campina Grande, Brazil, between October 2009 and April 2011 (ten monthly sampling efforts).

The paucity of biodiversity data did not allowed conducting any type of analyses and no particular pattern is discernible from the graphic (Figure 77).

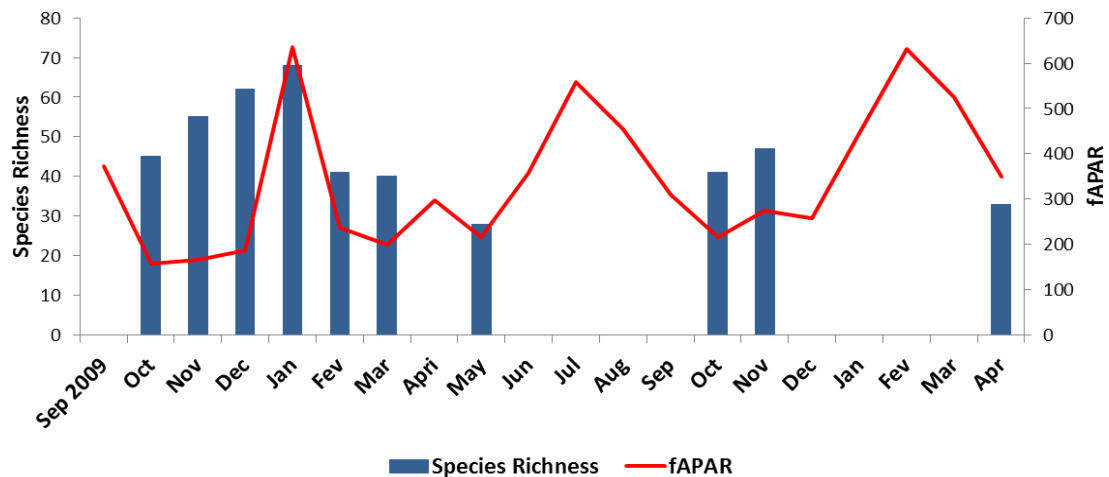


Figure 77: Plot of the second Caatinga trend analysis. Charts depict biodiversity measure (left Y axis) and fAPAR (right Y axis), per sampling date (X axis, from September 2009 to April 2011). Blue horizontal bars represent biodiversity data and red line fAPAR.

3.2.2.5 West Sudanian Savannah

Two data sources were also available for the West Sudanian Savannah trend analysis. The first was comprised of yearly number of aquatic birds (belonging to the Anatidae group, i.e. ducks, geese and swans) within the Djoudj National Park in Senegal, from 2002 to 2010 (Triplet et al. 2010) (Figure 78).

The fAPAR data was averaged annually and for the rainy season (roughly between May and November), since the peaks of primary productivity occur within this period as well. Also the maximum and minimum values of each year were extracted. As explained above, Spearman's rank order correlation

analyses were conducted between fAPAR and biodiversity data. Significance of results were tested by their probability values (p), considering a confidence level of $p < 0.05$.

Statistically significant correlations were not retrieved between fAPAR and bird data (data not shown). An example with annual and rainy season fAPAR (maximum and minimum were similar) are given in Figure 79, in which no explicit pattern is discernible.



Figure 78: Geographic location of the Djoudj National Park for the first West Sudanian Savannah trend data analysis.

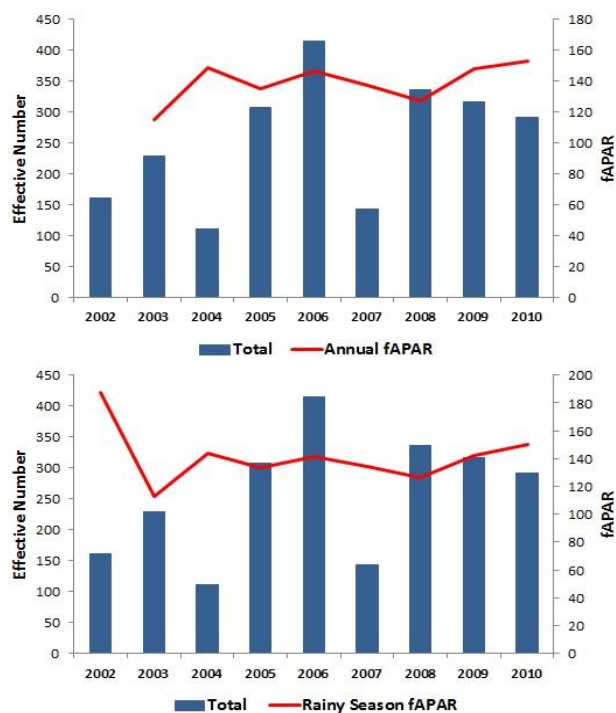


Figure 79: Plot of the first West Sudanian Savannah trend analysis. Charts depict biodiversity measure (left Y axis) and fAPAR (right Y axis), per sampling date (X axis, from 2002 to April 2010). Blue horizontal bars represent biodiversity data and red line fAPAR (top plot annual, lower plot rainy season).

The second biodiversity trend data was extracted from Konecky et al. (2009), consisting of relative abundance and species richness of small mammals, between December 2005 and September 2007 (only four sample dates), in four localities within and near the Niokolo-Koba National Park, Senegal (Figure 80). Biodiversity data was averaged between the four localities.

Again, the paucity of biodiversity data did not allowed us to conduct any type of analyses and no particular pattern is discernible from the graphics (Figure 81).

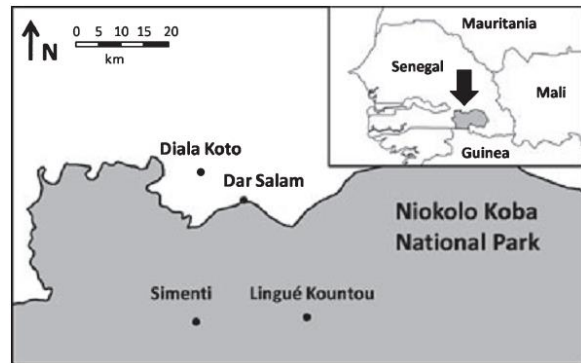


Figure 80: Geographic location of the Niokolo-Koba National Park for the second West Sudanian Savannah trend data analysis. Adapted from Konecky et al. 2009.

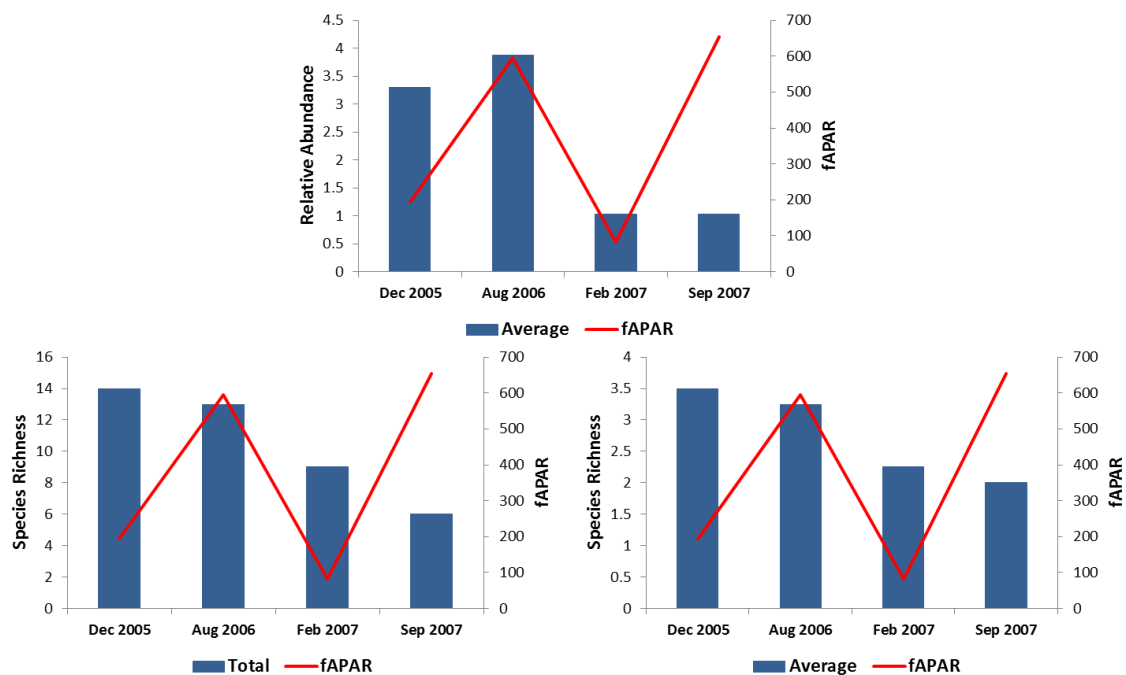


Figure 81: Plot of the second West Sudanian Savannah trend analysis. Charts depict biodiversity measure (left Y axis) and fAPAR (right Y axis), per sampling date (X axis, from December 2005 to September 2007). Blue horizontal bars represent biodiversity data (top plot relative abundance, low left plot total species richness and low right plot average species richness) and red line fAPAR.

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