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Concepts and evolution of systematic conservation planning in biodiversity conservation

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Abstract

Following Target 3 of the Kunming-Montreal Global Biodiversity Framework, nations across the world have agreed to protect 30% of the world's biodiversity by 2030. The protected sites established must represent all biodiversity, be well-connected and be governed equitably. Systematic conservation planning (SCP) is an approach to identifying where to establish protected areas that best represent the region's biodiversity. This essay will explore this approach and its evolution to understand the different tools and elements to consider in SCP. Furthermore, the disadvantages associated with the use of specific methods and the best approach to consider for a specific area will be discussed. In recent years, several concepts of conservation planning for site selection have been developed to determine locations for protected areas. These include several criteria: the biodiversity value of an area of interest, the threat level, the economic aspects (such as costs of protection) and the socio-ecological and socio-economic aspects. These concepts of conservation planning used for site selection will vary based on the conservation objectives and targets within the area of interest. Decision support tools have also evolved, from opportunistic site selection to systematic approaches, including complex algorithms and, most recently, the use of artificial intelligence (AI) tools. Therefore, systematic conservation planning is a powerful method used in the field of biodiversity conservation to reach targets, although some challenges might need to be taken into account in future assessments.

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Abbreviations

AI	Artificial Intelligence
CIA	Cumulative Impact Assessment
ILP	Integer Linear Programming
MPA	Marine Protected Area
MSP	Marine Spatial Planning
PA	Protected Area
RL	Reinforcement Learning
SCP	Systematic Conservation Planning

1. Introduction

1.1 Marine Spatial planning and systematic conservation planning evolution

Marine spatial planning (MSP) can be defined as a process for creating a strategic and integrated plan, including human activities both spatially and temporally, to manage the marine ecosystems and complete social, ecological or economic goals (Fernandes et al. 2018; Ehler et al. 2019). More precisely, MSP represents a way to design the use of the ocean space, including both the human activities present (such as fisheries, shipping, tourism, aquaculture, energy production, and marine mining) and the protection of the marine ecosystems present (Frazão Santos et al. 2019). In some countries, MSP first started approximately 40 years ago, thanks to the common interests of stakeholders to solve conflicts linked to the intensity of maritime use and the conservation of biodiversity. The goal for the stakeholders was to identify processes and tools to manage all marine activities (C. N. Ehler, 2021; Frazão Santos et al., 2019). However, the current aims of MSP tend to lean towards economic and social objectives associated with a need to facilitate multi-use planning of the marine space and in support of "blue growth" (Frazão Santos et al. 2019; Trouillet & Jay 2021).

In Europe, the EU Marine Spatial Planning Directive, established in 2014, is a framework for marine spatial planning to "support the sustainable development of seas and oceans" (European Union 2014). In the Baltic Sea, MSP started in 2005 with the *Balance project* (https://www.balance-eu.org/), which was a project aiming towards the design of marine management tools within the Baltic Sea based on marine spatial planning and co-operation. Although there is no Baltic-wide MSP, national marine spatial plans have been implemented in the Baltic Sea. The implementation of MSP plans differs between countries depending on their national strategy and objectives (Rodrigues & Milo-Dale 2022). In Sweden, there are three marine spatial plans: one for the Gulf of Bothnia, one for the Baltic Sea and one for the Skagerrak/Kattegat area. The goals of the marine spatial plans are specifically linked to their geographical locations, and the resources and uses linked to those locations (Swedish Agency Marine and Water Management 2022). For example, the Swedish plan within the Baltic Sea focuses on marine biodiversity conservation, due to the high biodiversity value of the area, and includes strategic environmental assessment and the establishment of a network of marine protected areas (MPA) (Rodrigues & Milo-Dale 2022; Swedish Agency Marine and Water Management 2022). Other countries in the Baltic Sea focus more on adopting a socio-economic or governance strategy (Rodrigues & Milo-Dale 2022).

The earliest efforts of conservation planning were guided by theoretical ecology, such as species-area curves (May 1975). Later, new algorithmic approaches linked to large amounts of data on biogeographic distributional information and socio-economic information started to be implemented (Sarkar et al. 2006). From that, systematic conservation planning (SCP) emerged, which differs from MSP as it settles a management plan with conservation as the main objective and socio-economic goals as secondary, compared to MSP, where the management of human activities, such as marine industries and energy, is the primary objective (Ehler 2018). SCP can be used as a process to fulfil the objectives of conservation, including the representation and persistence of biodiversity in the long term. To do that, conservation planning must combine different parameters, e.g. reserve location and design (e.g., size, connectivity, boundary length, and replication) when establishing protected areas (Margules & Pressey 2000).

Although some similarities can be noted between MSP and SCP, the two processes often take place in parallel, with few interactions between them (Reimer et al. 2023). This can be explained by the fact that conservation efforts have been implemented for many decades under established frameworks, which cannot be easily integrated into MSP processes (Trouillet & Jay 2021). Another noticeable difference is on the political side, where MSP might focus on sustainable blue growth, whereas SCP focuses mainly on conservation (which might be considered costly from a political point of view) (Schultz-Zehden et al. 2019). However, while the primary goals of MSP and SCP are different, some researchers argue that marine conservation should be part of the MSP process to make sure that economic and conservation activities are not in conflict and managed sustainably (Trouillet & Jay 2021).

1.2 Definition of Protected Areas and their ecological importance

Protected areas (PAs) are an important tool to counter the pressures linked to anthropogenic activities and climate change on habitats and species (Gray et al. 2016). According to the IUCN, a protected area can be defined as "a clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values" (Dudley 2008).

Likewise, marine protected areas (MPAs) are established to maintain marine biodiversity and ecosystem functions and services against anthropogenic pressures and climate change (Leenhardt et al. 2015). PAs and MPAs have become valuable tools in international conservation efforts. Indeed, international frameworks, such as Target 3 of the Kunming-Montreal Global Biodiversity Framework protocol, which aims to conserve 30% of land, waters and seas by 2030 (Convention on Biological Diversity 2022). The EU Biodiversity Strategy includes a target of 10% of strict protection and a minimum of 30% protection overall (European Commission. Directorate General for Environment. 2021). In addition, the recently adopted EU Nature Restoration Law stipulates a global target of at least 20% restoration of EU's land and marine areas by 2030 (European Union 2024).

Although the international community is interested in protected areas and marine protected areas through the development of these global targets and commitments, PAs and MPAs need to be carefully selected to achieve conservation benefits (Pressey & Tully 1994a; Mills et al. 2012). This is even more relevant in a context where: (i) there are spatial limitations concerning the protection of the marine and land areas; (ii) the resources allocated to conservation can be scarce in some cases;

and (iii) the conservation outcome needs to be efficient, in increasing biodiversity and reducing threats (Pressey & Tully 1994a). For these reasons, the use of processes such as conservation planning and systematic approaches can be useful in spatially allocating protected sites to be implemented in networks of PAs or MPAs, considering criteria such as the reserve size, connectivity and replication (Pressey & Tully 1994a; Margules & Pressey 2000).

1.3 Use of systematic conservation planning for marine protected areas

Some of the concepts and methods used in marine conservation planning have been adopted from those developed for terrestrial habitats (Hutchings & Lunney 2003). However, it is important to note that systematic conservation planning is used slightly differently between marine and terrestrial ecosystems, related to fundamental physical and biological differences. For example, the dynamism of oceanographic processes can influence the dispersal of marine species, and they may disperse either beyond or within delimited boundaries (Hutchings & Lunney 2003). Another difference is the complexity associated with the governance of seas and oceans, considered to be a global resource, typically with no private ownership (United Nations 1982). However, despite these differences, some aspects of conservation planning from terrestrial habitats can be adapted to marine ecosystems, as both systems are facing the same spatial challenges (constraints can be noted in terms of reserve selection and design criteria) and aiming for the same goals (Hutchings & Lunney 2003).

Within marine habitats, SCP can be used to: (i) delimit MPA boundaries to increase conservation outcomes or design zonation of an MPA (e.g., areas of no-take zones or areas with different regulations regarding fishery, tourism, etc.) (Vaughan & Agardy 2020) (ii) evaluate a network of MPAs and identify gaps in biodiversity conservation (Asaad et al. 2018) (iii) select new MPAs to design an efficient MPA network that will meet the biodiversity conservation and additional planning objectives (Álvarez-Romero et al. 2018) (iv) habitat restoration; and (v) evaluation of impacts from pressures and planning to avoid further ecological impacts (Moilanen et al. 2022).

2. Key principles in systematic conservation planning

2.1 Elements to include

Conservation planning requires the consideration of several elements. These elements will be described below. One is linked to the first stage of systematic conservation planning and consists of measuring and mapping biodiversity. **Biodiversity** can be divided into different levels depending on the spatial scale of the conservation planning analysis. Those levels include individuals, species, populations, communities, habitats and ecosystems (Margules & Pressey 2000). In addition to biodiversity, it is important to consider **ecological processes** across both spatial and temporal scales in conservation planning, as species distributions can vary with climate change and other pressures linked to human activities (Pressey et al. 2007; Van Der Biest et al. 2020). The selection of relevant ecological processes to include in SCP will vary depending on the knowledge and the data available for the specific geographical area (Pressey et al. 2007).

Connectivity and climate change are key features to consider in the design of MPAs, as they are tightly linked to the viability of populations and therefore the success of the MPAs (Magris et al. 2014). Connectivity also interacts with climate change, e.g. as climate change can affect larval dispersal pathways, spawning phenology, behaviour and mortality (Magris et al. 2014). Connectivity can facilitate the ability of populations to cope with the effects of climate change by facilitating recovery through the exchange of genotypes between populations (Magris et al., 2014; Munday et al., 2009).

Human activities and associated pressures should be considered at an early stage in the analysis, as they can have negative effects on habitats and ecosystems and thus decrease the resilience of biodiversity to future impacts (Dailianis et al. 2018). Those activities might vary depending on the area of interest, the spatial scale considered, and the relative spatial extent (such as offshore, coastline) (Korpinen et al. 2021). In the marine environment, human-induced pressures include for instance, increased sea surface temperature, noise disturbance, eutrophication, physical disturbance, and activities such as fisheries, offshore wind farms, military activities and tourism (Korpinen et al. 2021). For example, in Europe, the extraction of resources by fisheries is the most frequently documented pressure in marine waters (Dailianis et al. 2018).

Ecosystem services represent the functions of the ecosystem that are used for human well-being (Van Der Biest et al. 2020). Marine ecosystem services can provide direct benefits to society, such as fish harvests, or indirect benefits, such as pollution control or carbon sequestration (Barbier 2017). Conservation of biodiversity by the establishment of MPAs can allow the preservation of marine ecosystem services (Leenhardt et al. 2015). The protection of the marine environment by MPAs can, for example, strengthen the ability of coastal ecosystems to produce goods and services for local communities, as well as enabling sustainable exploitation of fisheries resources (Leenhardt et al. 2015; Costello 2024). Accordingly, it might be important to consider ecosystem services in conservation planning, including both biotic and abiotic processes, as ecosystem services are driven by both biological and physical processes (Van Der Biest et al. 2020).

Finally, the goal of systematic conservation planning is to achieve the conservation targets at the least possible cost (Naidoo et al. 2006; Watson et al. 2011). Therefore, the **costs of protection** should be included in SCP analyses, as they may have a large impact on the selection of priority conservation areas and the possibility of delivering plans that are feasible and politically acceptable (Gissi et al. 2018).

2.2 Spatial scales and their impacts on the analyses

In addition to the elements outlined above, it is important to consider different **spatial scales** in marine conservation analyses. Ecological processes include a wide array of hierarchical spatial scales, which range from local to global (intercontinental) (Huber et al. 2010). Scales can be based on biological data and will affect the systematic selection algorithms by influencing the distribution, number, and total area of the selected sites (Warman et al. 2004). A spatial scale might need to be included for the factors that are part of the planning processes, such as the extent and boundaries of the planning area, the data resolution according to the area and the resolution of the planning units for the assessment, and the design and implementation of MPA networks (Mills et al. 2010).

At larger scales, such as global, EU, pan-Baltic, or national, priority areas for conservation efforts can be defined. At local scales, prioritisation analyses can guide the placement of protected areas within networks (Pressey et al. 1993). For example, a study focusing on systematic conservation planning for Finnish butterflies performed at different spatial scales showed that the conservation objectives and optimisation methods could be different depending on the scale chosen (along with the planning units, the species data and the model variables) (Cabeza et al. 2010). The Finnish study shows that a national-scale analysis can allow the identification of conservation priorities for species representation (i.e.: the analysis identified two different areas important for butterfly in Finland, which will be relevant to protect) based on a species distribution and landscape classification, whereas a local scale will support the finding of specific sites that can benefit population persistence (Cabeza et al. 2010). Conservation planning studies tend to focus on large spatial scales (such as global or national), and the choice of scale is often based on the availability of the data (Cabeza et al. 2010). However, certain features, such as connectivity, can vary over large spatial scales. Therefore, local conservation planning might miss certain aspects of connectivity, as it will focus narrowly on site-specific objectives. For example, a local conservation plan on a specific nature reserve might miss the movement of certain species moving between several nature reserves. Similarly, gene flow can occur over a large temporal and spatial scale, meaning that short-term conservation planning could miss this aspect.

Consequently, several factors influence the choice of the spatial scale, which can be guided by the ecological, social, or economic strategy of conservation plans (Mills et al. 2010). However, planners and managers might have to work on various scales to design effective networks of protected areas, as there might be variability in the scales used depending on the conservation objectives (Warman et al., 2004; Huber et al., 2010). For instance, with connectivity, managers could include a local scale by ensuring the protection of the local habitats. On a bigger scale, regionally, they could ensure that the connectivity between the sites is maintained. Finally, on a global scale, their work could involve species distance movements and migration routes.

2.3 Stakeholder involvement

Another element to include in systematic conservation planning is **stakeholder involvement**. A stakeholder can be defined as a group of people, organised or not, that share a common interest in a specific system (Brown et al. 2016). In marine conservation, many stakeholders are not involved in conservation directly, but will be affected by conservation. Therefore, stakeholders should be involved in the process. Those include representatives from the government or industries, such as fisheries, shipping, oil and gas companies, offshore wind farming, or members of indigenous communities. Other stakeholders include research scientists, non-governmental organisations, or volunteers (Ison et al. 2021).

Relevant stakeholders' involvement is important to consider in marine conservation to provide effective marine conservation planning and management, as this will allow efficient communication and collaboration between the stakeholders and avoid conflicts at a later stage (Brown et al. 2016). They can be involved in different stages of conservation planning (or all stages), such as (i) the planning phase, where the objectives and priorities are set; (ii) the evaluation phase, where conservation planning options and outcomes are evaluated; (iii) the implementation phase, where the conservation planning plan is implemented and;

(iv) the post-implementation phase, where the effectiveness of the conservation plan is evaluated (Pomeroy & Douvere 2008).

3. Concepts of conservation planning for site selection

There are different concepts in conservation planning used in the selection of sites that can be divided into four main categories: the biodiversity value, the threats that potential conservation values are facing, the economic aspects (including costs of protection) and the social aspects of reserve selection (e.g., equity, gender, religion, human rights) (Margules & Pressey 2000; Kukkala & Moilanen 2013; Sacre et al. 2019).

3.1 Biodiversity value

The biodiversity value of a potential site can be evaluated based on several concepts in conservation planning, which are described below.

3.1.1 Species richness

The selection of MPAs can be determined based on species richness, which represents the sites with the highest number of species or threatened species within a specific area (Astudillo-Scalia & Albuquerque 2020). Species richness is an indicator often used in conservation planning to identify important areas to prioritise conservation efforts (Fleishman et al. 2006; Astudillo-Scalia & Albuquerque 2020). However, there might be several limitations to the utilization of species richness alone in conservation planning, as it does not provide information about species-specific functional roles in the ecosystem, species composition, life history or distribution patterns. Therefore, there is no information about the dynamic processes present (such as species interaction or ecological connectivity, for instance), and potential biases might impact site selection process if only species richness is considered (Fleishman et al. 2006). Consequently, while species richness is an important concept in MPA selection, it may be advantageous to combine it with complementary approaches, such as connectivity and ecological processes (Fleishman et al. 2006).

3.1.2 Complementarity

A concept stemming from species richness and species composition is complementarity, which includes selecting sites that are complementary to each other. For example, different MPAs should include species and habitats that together increase the representation of overall biodiversity in the whole region (Watson et al. 2011; Kukkala & Moilanen 2013). Complementarity can be defined as the number of underrepresented biodiversity features that a new site is adding to the overall network (Margules & Pressey 2000). In other words, complementarity allows avoiding duplication of conservation efforts, so that new areas selected during planning complement the previously chosen protected areas (Linke et al. 2011).

3.1.3 Representation

Representation is the occurrence of a variety of biodiversity features (such as species occurrence) that are protected within the selected sites in the region (Kukkala & Moilanen 2013). It is a conservation target or goal that needs to be satisfied following a minimum cost to lead to representativeness of the network and maximise efficiency of protection of biodiversity (Kukkala & Moilanen 2013). However, this approach focuses mainly on maximising the number of species protected without including the long-term persistence of species representation or the specific need for protection for the biodiversity features (e.g., threatened species will require a higher protection coverage) identified (Pressey et al. 2002; Cabeza et al. 2010). Consequently, additional factors may need to be taken into account in conservation planning analyses, such as connectivity or habitat suitability, to better ensure species persistence in the future (Cabeza et al. 2010).

3.1.4 Adequacy and persistence

Adequacy refers to the long-term conservation of the ecological viability and coherence of populations, species, and communities to ensure their persistence. This concept is used in systematic conservation plans and can be defined as a target percentage necessary to ensure the persistence of each species in the future (Possingham et al. 2006; Watson et al. 2011; Kukkala & Moilanen 2013). Those target percentages can be determined by scientific knowledge regarding species ecology and ecological processes (Watson et al. 2011; Reside et al. 2018).

Persistence (i.e., long-term survival of species, habitats or ecosystems) is used in conservation planning and often includes consideration of climate change and other area-use changes in the future (Reside et al. 2018). To achieve persistence, continuous functional integrity of biological communities and processes, such as species interactions or energy flow, is required. In addition, it is important to maintain ecological connectivity to enable the persistence of populations, species, communities, and ecosystems (Magris et al. 2014; Beger et al. 2022). It has also been suggested that additional factors granting the persistence of biodiversity features in the future should be included, such as sufficient area protection, habitat quality and genetic diversity (Beger et al. 2022). The inclusion of these factors will allow the planners to design a resistant network of protected areas.

3.1.5 Irreplaceability

Irreplaceability represents the relative importance of a specific site or its degree of uniqueness (Asaad et al. 2018), with the highest values corresponding to only a few or no similar or identical sites (Pressey et al. 2009). This concept is also linked to vulnerability, as it is used to prioritise sites that need urgent conservation actions, i.e. those that are considered vulnerable (due to high risk of being transformed because of high levels of threats impacting the survival, abundance, or development of species or the ecological community) (Wilson et al. 2005; Kukkala & Moilanen 2013). Vulnerability includes three dimensions: exposure (refers to the probability of a threat affecting a specific area over time), intensity (refers to the magnitude, frequency, and duration of a threat), and impact on species distribution, abundance, or likelihood of persistence (Wilson et al. 2005). To make conservation decisions, it has been suggested to plot the selected sites on two axes with irreplaceability against vulnerability. The areas with the highest values for both concepts are likely to have the highest conservation priority (Margules & Pressey 2000).

3.1.6 Flexibility

Flexibility is a relevant concept to be included in systematic conservation planning. It is a method based on the principle that biodiversity features have a patchy distribution across the seascape. Therefore, it is important to include different types of scenarios in SCP, with different selection procedures, for the stakeholders to have multiple options that suit different objectives (Pressey et al. 1993; Stewart et al. 2003; Kukkala & Moilanen 2013). Having alternative planning scenarios with different objectives may allow the planners to find the best option that maximises the representation of biodiversity features while reducing the costs of protection (Pressey et al. 1993).

3.2 Threats

It is important to consider threats to achieve appropriate conservation targets, because different pressures stemming from human activities may have impacts on biodiversity that vary through time and space (Virtanen et al., 2022; Kukkala & Moilanen, 2013). Having a good understanding of key threats will help conservation decisions by highlighting conservation priority areas (Margules & Pressey 2000). Importantly, two threat prioritization strategies can be discussed: (i) prioritization of "frontier" areas, which are areas subject to high levels of threats, and (ii) prioritization of "pristine" areas, which are areas with low levels of threat but that may be threatened in the future (Sacre et al. 2019). The decision on which prioritization strategies are most efficient for conservation outcomes can depend on different factors, such as costs, biodiversity value, the potential of biodiversity recovery within an area, threat dynamics over time, and the timeframe decided for the conservation objectives to be fulfilled (Sacre et al. 2019).

3.3 Economic aspects

Several types of costs can be considered, such as management, monitoring, and opportunity costs (Mazor et al. 2014). Opportunity costs are the foregone economic losses following the implementation of protection within an area (Appolloni et al. 2018). The most common opportunity cost in marine conservation planning is related to prohibiting fishing within an MPA, which may have an impact on the profits previously generated in the area before its designation as a closed/no-take MPA. Indeed, in the case of no-take zones, which are marine protected areas where fishing is not allowed within the area boundaries, this may lead to economic losses both for commercial and recreational fisheries because of lost harvest and profits (Bostedt et al. 2020). However, from a long-term perspective, fishing limitations in MPAs might have long-term benefits on the fishing economy through an increase in catches because of the spillover of adult fish and an increase in larval supply, in which case the opportunity costs will be lowered (Bostedt et al. 2020). Opportunity costs also include commercial or recreational activities other than fishing, such as tourism, recreation or aquaculture, for instance. In systematic conservation planning, opportunity costs are calculated as the highest economic value of extractive and exploitation activities when no form of protection is set (Appolloni et al. 2018).

Management costs represent the costs linked to the management of a conservation plan of a network of protected areas, and can vary depending on the location (Naidoo et al. 2006). Monitoring costs are costs linked to the tools used to obtain information on species, their threats, and their responses to other measures of biodiversity, conservation and management plans in the long term (Buxton et al. 2020).

There is a trade-off between the costs of protection versus the benefits related to the conservation of an area. The designation of MPAs benefits the marine habitats and ensures their functionality, the restoration of overexploited fish stocks and the provisioning of ecosystem services (Galparsoro & Borja 2021; Vigo et al. 2024). On the other hand, MPAs can also lead to a displacement of fishing effort, which might increase fishing pressures somewhere else, leading to more competition among fishers outside MPAs. Furthermore, local communities' livelihoods might be negatively affected by the implementation of protected areas (Abachebsa 2017). SCP can be used to balance the importance of MPAs to biodiversity conservation and the impacts of protection on human activities (Brown et al. 2015).

Finally, it is important to note the challenge associated with the estimation of the costs of protection in the marine realm, being a common open resource unlike most terrestrial ecosystems. Indeed, the marine environment is transboundary, where the resources and the different pressures, with varying cumulative impacts (such as pollution), are moving freely in the spatial area of interest. This raises the uncertainties of the conservation planning analyses (Börger et al. 2016).

3.4 Social aspects

The last dimensions to consider in systematic conservation planning are the socio-ecological and socio-economic aspects. For instance, the implementation of a marine protected area with its associated rules and regulations may have some impact on the livelihood of the communities living close to this area because of the restrictions set on resource extraction. This may lead to food insecurity and impoverishment of resource users (Cánovas-Molina & García-Frapolli, 2020; Moshy et al., 2015). A concrete example of this conflict is the case of the Mafia Island Marine Park in Tanzania, where conflicts were raised between stakeholders (Mcclanahan et al., 2008). This MPA was designated and regulated without consideration of the local communities and the impacts on their livelihoods. As a result, several fishing grounds were closed and restrictions on fishing gear were implemented, preventing the resource users from the area where they needed to maintain a viable livelihood (Moshy et al. 2015).

The lack of involvement of local communities in protected area management can also lead to conflicts (Abachebsa 2017). Returning to the example of Mafia Island, communication between the MPA manager and the local communities decreased, and violent enforcement methods were implemented, which inhibited good relationships between stakeholders and managers (Moshy et al. 2015). Indeed, if resource users are not engaged or if they do not agree with the plan, they might not comply with the set rules of protection, leading to poor success in the management implementation.

In other cases, the establishment of no-take zones contributes to sustainable fisheries and can improve the livelihood of local communities, supporting economic activities such as fisheries and aquaculture. Furthermore, ecosystem services such as cultural services are beneficial and are important for recreation and leisure (Pegorelli et al. 2024). However, to improve the involvement of the local communities, stakeholders must be engaged from the beginning when conservation and management plans are designed (Brown et al. 2016). For instance, during the designation of an MPA, managers, local communities, NGOs, researchers, and resource users must be present and design the plans together. (Young et al. 2013; Brown et al. 2016; Adams et al. 2019; Vaughan & Agardy 2020). Therefore, a shift towards a bottom-up participatory approach, where the local communities and the resource users are leading the management and conservation processes, is emphasised by NGOs and scientists (Oyanedel et al. 2016). In addition, the use of a bottom-up participatory approach may reduce potential conflicts between resource users and managers (Mcclanahan et al., 2008). Stakeholders' participation may also raise understanding about the importance of the conservation of ecosystems and species. Furthermore, the inclusion of local communities will allow for the incorporation of local knowledge and, therefore, the collection of the best available data both from the scientific community and from the local communities (Day 2017).

However, it is important to note that social aspects might vary depending on the area of interest, and there is no universal method. Every situation may be different depending on the MPAs' management plans and objectives, and the local political, economic, and social context of the area of interest (Day 2017).

4. Evolution of conservation planning and decision support tools

The methods for prioritising protected areas are evolving from opportunistic to systematic and more scientific-based approaches (Stewart et al. 2003; Watson et al. 2011). In this section, the evolution of conservation planning and decision support tools, starting from an opportunistic selection to the use of detailed and expert tools, will be described.

4.1 Opportunistic reserve selection

The first scientific efforts to design protected area networks were based on the theory of island biogeography (MacArthur & Wilson, 1967). Protected sites were accordingly viewed as isolated islands surrounded by areas affected by anthropogenic pressures. This theory of efficient protection suggests that larger protected areas harbour more species and that bigger MPAs therefore likely protect more species (Watson et al. 2011). Around the 1980s, areas were chosen by conservationists, especially in regions that were easy to protect for political and economic reasons, such as those with limited human activities (Margules & Usher 1981; Pressey & Tully 1994a). They could also be chosen through discussion with community stakeholders (Hansen et al. 2011). The species chosen to be protected were based on simple criteria such as species richness or the number of endemic species (Watson et al. 2011). This approach is called opportunistic as the MPAs' locations are chosen in opportunity areas where they will be easily implemented and enforced (Hansen et al. 2011).

There are no specific tools used in opportunistic approaches, and the most common method consists of identifying site locations through stakeholder discussions (e.g., with scientists, managers, governments, or local communities).

Criteria for reserve selection may vary depending on the goals within the area of interest (Hansen et al. 2011). This approach is known to be cheap and easy to enforce, as it does not require a lot of resources and knowledge (in terms of data acquisition, for instance), and it can be fast to implement in a context where biodiversity loss happens at a fast rate (Hansen et al. 2011). However, opportunistic approaches are associated with several downsides. The major downside of this approach is that it does not consider the concepts described in the previous section, such as representation, irreplaceability, connectivity, complementarity, etc. Indeed, the number of species present in a single area should not be the only factor to consider when selecting a new protected area (Pressey & Tully 1994b; Watson et al. 2011). Furthermore, protected areas should be located strategically in areas with high concentrations of threatened species or high pressures, depending on the conservation objectives of the area of interest, instead of areas that minimise conflicts with resource users (Venter et al. 2018). From opportunistic approaches, systematic conservation planning emerged as a more efficient means to achieve conservation objectives (Hansen et al. 2011; Watson et al. 2011).

4.2 Systematic approaches to identify and prioritize sites of conservation

In comparison to opportunistic approaches, systematic approaches include selection algorithms that are used to select potential new protected sites or expand the existing network of protected areas. Those new areas to be protected will be selected based on conservation targets set for the designated area and on the principles of SCP described in the previous section (Galparsoro & Borja 2021). Furthermore, the selection algorithms can generate several alternative networks, all meeting the conservation objectives (Hansen et al. 2011).

In systematic approaches, key biodiversity features (such as species or habitats) are targeted as a priority (Hansen et al. 2011; Watson et al. 2011). Furthermore, the area of interest is divided into a raster, which will be split into delineated areas called planning units, referring to areas of interest that could become potential protected areas (Hanson et al. 2024).

Systematic approaches commonly include several stages, such as: (i) identification of conservation goals and stakeholders for the area of interest, followed by; (ii) data collection (biodiversity data, socio-economic data); (iii) setting the conservation objectives and targets, and; (iv) assessment of the area according to the targets sets and prioritisation of conservation areas with the implementation of regulations in the long term to meet the conservation objectives (Pressey & Bottrill 2008). Some of the tools used to select potential protected sites to networks of protected areas will be described below.

4.2.1 Simulated annealing (Marxan)

The first selection algorithm that is worth mentioning is **simulated annealing.** This algorithm can be applied using the Marxan decision support software (Marxan 2022). Marxan is a decision-support tool used for the identification of new protected areas according to set ecological, social, and economic goals(Fernandes et al. 2018). Its goal is to find a list of solutions for protected area networks that meet the targets set by planners at a minimal cost (Serra et al. 2020). The most commonly used algorithm in Marxan is simulated annealing, a repetitive and stochastic algorithm that is used to identify a set of multiple solutions represented as combinations of protected areas (Serra et al. 2020). This algorithm is a heuristic algorithm, which generates solutions that are "near-optimal", which means that the algorithm will provide a range of "good" options for conservation planners and the stakeholders to take into account (Serra et al. 2020). However, the main disadvantage of using this approach is that the solution quality is unknown, as there are no guidelines to parameterize the algorithm (Beyer et al. 2016; Esfandeh & Kaboli 2019; Schuster et al. 2019).

4.2.2 Ranking (Zonation)

Another selection algorithm used in systematic approaches is the **zonation algorithm**, based on spatial priority ranking, starts with the entire seascape, divided into cells. Afterwards, based on the criteria with the least economic loss of conservation value, cells are removed iteratively from the grid, producing a hierarchical prioritization across the area of interest (Moilanen et al. 2005;

Moilanen 2007; Virtanen et al. 2018). In addition, the features used in the analysis are weighted. The weight of a feature will vary depending on, for example, the conservation status, economic value, and phylogenetic uniqueness of species and habitats (Virtanen et al. 2018). The choice of cells that will be removed from the grid will depend on the conservation targets set for the species. This algorithm produces a ranking of different conservation priorities through the seascape, aiming for a maximum coverage type solution, where the goal is to reach as many of the conservation targets set as possible, with a limited budget available (Moilanen 2007). Given that data is available, Zonation can also include connectivity, ecosystem services, costs, and threats in the analyses (Virtanen et al. 2018).

4.2.3 Integer linear programming (Gurobi and Prioritizr)

Furthermore, **integer linear programming** (ILP) is an optimisation approach used to minimize or maximize a mathematical function. In conservation planning, this function may consist of an objective function, e.g. decreasing species representation according to a budget cost to improve feature representation if the conservation funding is limited (Beyer et al. 2016; Hanson et al. 2024). In comparison to simulated annealing, ILP generates faster and higher-quality solutions, or solutions that are within the defined shortfall of the optimum (Beyer et al. 2016; Schuster et al. 2019). Furthermore, another benefit of using ILP is that it is not necessary to set certain parameters, such as the number of iterations, the penalty factors, or the number of restarts, for instance. This step is time-consuming, especially in the case of large/high-resolution datasets (Schuster et al. 2019). However, integer linear programming cannot solve non-linear or complex problems, as this approach is used principally for simple problems (Moilanen 2008; Beyer et al. 2016). In such cases, nonlinear/stochastic problems could be more beneficial to use (Moilanen 2008; Hanson et al. 2024). However, certain common conservation planning problems can be linearized to use ILP and find optimal solutions (Beyer et al. 2016).

ILP can be performed using the open-source *PrioritizR* package software, which utilizes the Gurobi solver function. A solver function allows defining the software and the settings that will be used in the analysis (such as the maximum running time, for instance) (Hanson et al. 2024).

4.2.4 C-Plan

In addition, another support tool used in systematic conservation planning is **C-Plan**, a decision-support tool that uses a statistical approach to set estimates of irreplaceability. For doing this, the tool estimates the number of sites that are needed to meet the targets set. Afterwards, for each site selected, the irreplaceability is predicted and recomputed at each selection run until all the targets are met. The first site selected by the algorithm will have the highest irreplaceability value (Carwardine et al. 2007). As a decision support tool, C-Plan provides fast information on site selection and can facilitate discussion amongst stakeholders (Pressey et al. 2009). The tool provides a fast method to estimate irreplaceability values and is beneficial for exploration of the area of interest. However, this approach is less efficient for the simultaneous accomplishment of various conservation objectives than Marxan, which can estimate irreplaceability for multiple objectives (Carwardine et al. 2007).

4.2.5 Artificial Intelligence (AI) approaches

Last but not least, the last approach worth mentioning is the use of **artificial intelligence**, or AI, in conservation planning, as it allows for the enhancement of the capacity of data analysis (Ullah et al. 2025). For instance, using AI, a reinforcement learning (RL) algorithm has been designed, especially for SCP, based on species distribution stimulations through time in response to threats (Silvestro et al. 2022). The RL algorithm includes time-dependent variables, which are added to the model, such as species richness, population density, economic value, phylogenetic diversity, anthropogenic disturbance, species rank abundance and climate change. Furthermore, Silvestro and his colleagues established a framework called CAPTAIN (Conservation Area Prioritization through Artificial Intelligence) to enhance either a static policy (i.e., where the funds are spent directly) or a conservation policy where conservation objectives and plans are developed over time, which can apply to managers establishing policies in protected areas (Silvestro et al. 2022). After training the model with the RL algorithm, it can be used in conservation prioritization using both simulated and empirical data (Silvestro et al. 2022). The utilisation of AI allows improving the data collection efficacy as well as the analysis of the data, as AI can handle large amounts of environmental data, allowing it to identify solutions and take conservation actions and decisions faster (Ullah et al. 2025).

4.2.6 Integration of cumulative impact assessment within systematic conservation prioritisation

A range of anthropogenic pressures strongly affect the marine environment through direct (e.g. resource extraction and physical disturbance such as noise pollution) and indirect uses (e.g. pollution from land-based activities). All these pressures will have a cumulative impact on marine ecosystems (Halpern et al. 2008; Whitehead et al. 2017). Therefore, including pressures in SCP using information about the spatial distribution of human activities is important to consider. For this purpose, cumulative impact assessments, or CIAs, can be used. CIA is a function including: (i) human pressures as intensity maps, (ii) ecosystem component maps, and (iii) sensitivity indices measuring how sensitive each specific ecosystem component is to each human pressure (Hammar et al. 2020). Therefore, the use of CIA in conservation planning is relevant as it allows for the inclusion of an accumulation of different kinds of stressors on biodiversity (Whitehead et al. 2017; Hammar et al. 2020). However, there are a few general shortcomings of CIA, i.e. that the model does not include connectivity, species-specific data, or food-web interactions (Hammar et al. 2020). A way to counter this limitation would be to use CIA in combination with the conservation planning approaches described above (Whitehead et al. 2017). Another shortcoming is that only a few CIA studies include all the pressures present in an area of interest in the assessments, leading to a bias towards the most common pressures in the area (such as pollution and noise in the Baltic Sea) (Korpinen & Anderson 2016). Other studies have attempted to aggregate similar pressures coming from similar areas in their assessments to facilitate the analyses. For instance, Korpinen and his colleagues combined pressures taking place in the coastal area together (such as “species disturbance by human presence” and “hydrographical changes”), separated from the pressures taking place in the continental shelf area (such as “input of nutrients” and “physical disturbance”)(Korpinen & Andersen 2016; Korpinen et al. 2021).

5. Conclusion

To conclude, systematic conservation planning is a useful approach to reach multiple biodiversity conservation targets simultaneously, considering, for instance, social and economic aspects in a systematic and transparent way. This is important, especially with the growing interest of international communities to reach 30 % biodiversity coverage by 2030. Systematic conservation planning support tools have evolved to become more efficient, faster and to include different concepts of conservation planning, such as ecosystem services, climate change, connectivity, and different aspects of biodiversity. Despite these improvements, tools in SCP still face some challenges. Among those challenges is the need to establish solutions that include climate change in the planning phase, especially since ecosystems change over time. Integrating the dynamic nature of ecosystems is important and should be captured by the network of MPAs to protect both the current and future important areas and be resilient to changes. Other knowledge gaps include data availability and data sharing between different sectors or countries. Furthermore, data can be available at different scales, and there is a need to agree on the most relevant scale to perform the analysis. Another knowledge gap that is worth mentioning is the inclusion of social and cultural dimensions in the analysis and the need to collaborate across fields. Thus, incorporating those challenges could potentially enhance the tools and methods used in systematic conservation planning, and I will most probably face those knowledge gaps during my PhD as well.

References

Abachebsa, A.M. (2017). Review on Impacts of Protected Area on Local Communities' livelihoods in Ethiopia. *Journal of Resource Development and Management*, Vol.39

Adams, V.M., Mills, M., Weeks, R., Segan, D.B., Pressey, R.L., Gurney, G.G., Groves, C., Davis, F.W. & Álvarez-Romero, J.G. (2019). Implementation strategies for systematic conservation planning. *Ambio*, 48 (2), 139–152. https://doi.org/10.1007/s13280-018-1067-2

Álvarez-Romero, J.G., Mills, M., Adams, V.M., Gurney, G.G., Pressey, R.L., Weeks, R., Ban, N.C., Cheok, J., Davies, T.E., Day, J.C., Hamel, M.A., Leslie, H.M., Magris, R.A. & Storlie, C.J. (2018). Research advances and gaps in marine planning: towards a global database in systematic conservation planning. *Biological Conservation*, 227, 369–382. https://doi.org/10.1016/j.biocon.2018.06.027

Appolloni, L., Sandulli, R., Vetrano, G. & Russo, G.F. (2018). A new approach to assess marine opportunity costs and monetary values-in-use for spatial planning and conservation; the case study of Gulf of Naples, Mediterranean Sea, Italy. *Ocean & Coastal Management*, 152, 135–144. https://doi.org/10.1016/j.ocecoaman.2017.11.023

Asaad, I., Lundquist, C.J., Erdmann, M.V., Van Hooidonk, R. & Costello, M.J. (2018). Designating Spatial Priorities for Marine Biodiversity Conservation in the Coral Triangle. *Frontiers in Marine Science*, 5, 400. https://doi.org/10.3389/fmars.2018.00400

Astudillo-Scalia, Y. & Albuquerque, F. (2020). Why should we reconsider using species richness in spatial conservation prioritization? *Biodiversity and Conservation*, 29 (6), 2055–2067. https://doi.org/10.1007/s10531-020-01960-4

Barbier, E.B. (2017). Marine ecosystem services. *Current Biology*, 27 (11), R507–R510. https://doi.org/10.1016/j.cub.2017.03.020

Beger, M., Metaxas, A., Balbar, A.C., McGowan, J.A., Daigle, R., Kuempel, C.D., Treml, E.A. & Possingham, H.P. (2022). Demystifying ecological connectivity for actionable spatial conservation planning. *Trends in Ecology & Evolution*, 37 (12), 1079–1091. https://doi.org/10.1016/j.tree.2022.09.002

Beyer, H.L., Dujardin, Y., Watts, M.E. & Possingham, H.P. (2016). Solving conservation planning problems with integer linear programming. *Ecological Modelling*, 328, 14–22. https://doi.org/10.1016/j.ecolmodel.2016.02.005

Börger, T., Broszeit, S., Ahtiainen, H., Atkins, J.P., Burdon, D., Luisetti, T., Murillas, A., Oinonen, S., Paltriguera, L., Roberts, L., Uyarra, M.C. & Austen, M.C. (2016).

Assessing Costs and Benefits of Measures to Achieve Good Environmental Status in European Regional Seas: Challenges, Opportunities, and Lessons Learnt. *Frontiers in Marine Science*, 3. https://doi.org/10.3389/fmars.2016.00192

Bostedt, G., Berkström, C., Brännlund, R., Carlén, O., Florin, A.-B., Persson, L. & Bergström, U. (2020). Benefits and costs of two temporary no-take zones. *Marine Policy*, 117, 103883. https://doi.org/10.1016/j.marpol.2020.103883

Brown, C.J., White, C., Beger, M., Grantham, H.S., Halpern, B.S., Klein, C.J., Mumby, P.J., Tulloch, V.J.D., Ruckelshaus, M. & Possingham, H.P. (2015). Fisheries and biodiversity benefits of using static versus dynamic models for designing marine reserve networks. *Ecosphere*, 6 (10), 1–14. https://doi.org/10.1890/ES14-00429.1

Brown, G., Strickland-Munro, J., Kobryn, H. & Moore, S.A. (2016). Stakeholder analysis for marine conservation planning using public participation GIS. *Applied Geography*, 67, 77–93. https://doi.org/10.1016/j.apgeog.2015.12.004

Buxton, R.T., Avery-Gomm, S., Lin, H.-Y., Smith, P.A., Cooke, S.J. & Bennett, J.R. (2020). Half of resources in threatened species conservation plans are allocated to research and monitoring. *Nature Communications*, 11 (1), 4668. https://doi.org/10.1038/s41467-020-18486-6

Cabeza, M., Arponen, A., Jäättelä, L., Kujala, H., Van Teeffelen, A. & Hanski, I. (2010). Conservation planning with insects at three different spatial scales. *Ecography*, 33 (1), 54–63. https://doi.org/10.1111/j.1600-0587.2009.06040.x

Cánovas-Molina, A. & García-Frapolli, E. (2020). Untangling worldwide conflicts in marine protected areas: Five lessons from the five continents. *Marine Policy*, 121, 104185. https://doi.org/10.1016/j.marpol.2020.104185

Carwardine, J., Rochester, W.A., Richardson, K.S., Williams, K.J., Pressey, R.L. & Possingham, H.P. (2007). Conservation planning with irreplaceability: does the method matter? *Biodiversity and Conservation*, 16 (1), 245–258. https://doi.org/10.1007/s10531-006-9055-4

Convention on Biological Diversity (2022). *Kunming-Montreal Global Biodiversity Framework*. CBD - UNEP. chrome-extension://efaidnbmnnnibpcajpcglclefindmkaj/https://www.cbd.int/doc/decisions/cop-15/cop-15-dec-04-en.pdf

Costello, M.J. (2024). Evidence of economic benefits from marine protected areas. *Scientia Marina*, 88 (1), e080. https://doi.org/10.3989/scimar.05417.080

Dailianis, T., Smith, C.J., Papadopoulou, N., Gerovasileiou, V., Sevastou, K., Bekkby, T., Bilan, M., Billett, D., Boström, C., Carreiro-Silva, M., Danovaro, R., Fraschetti, S., Gagnon, K., Gambi, C., Grehan, A., Kipson, S., Kotta, J., McOwen, C.J., Morato, T., Ojaveer, H., Pham, C.K. & Scrimgeour, R. (2018). Human activities and resultant pressures on key European marine habitats: An analysis of mapped resources. *Marine Policy*, 98, 1–10. https://doi.org/10.1016/j.marpol.2018.08.038

Day, J.C. (2017). Effective Public Participation is Fundamental for Marine Conservation—Lessons from a Large-Scale MPA. *Coastal Management*, 45 (6), 470–486. https://doi.org/10.1080/08920753.2017.1373452

Dudley, N. (2008). Guidelines for Applying Protected Area Management Categories. IUCN.

https://www.iucn.org/sites/dev/files/import/downloads/iucn_assignment_1.pdf [2024-09-12]

Ehler, C., Zaucha, J. & Gee, K. (2019). Maritime/Marine Spatial Planning at the Interface of Research and Practice. In: Zaucha, J. & Gee, K. (eds) *Maritime Spatial Planning*. Springer International Publishing. 1–21. https://doi.org/10.1007/978-3-319-98696-8_1

Ehler, C.N. (2018). Marine spatial palnning: An idea whose time has come. In: *Offshore energy and marine spatial planning*. 1st Edition. 12.

Ehler, C.N. (2021). Two decades of progress in Marine Spatial Planning. *Marine Policy*, 132, 104134. https://doi.org/10.1016/j.marpol.2020.104134

Esfandeh, S. & Kaboli, M. (2019). Using simulated annealing optimization algorithm for prioritizing protected areas in Alborz province, Iran. *Environmental Nanotechnology, Monitoring & Management*, 11, 100211. https://doi.org/10.1016/j.enmm.2019.100211

European Commission. Directorate General for Environment. (2021). *EU biodiversity strategy for 2030: bringing nature back into our lives.* Publications Office. https://data.europa.eu/doi/10.2779/677548 [2024-09-10]

European Union (2014). Directive 2014/89/EU of the European Parliament and of the Council of 23 July 2014 establishing a framework for maritime spatial planning. European Union. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex%3A32014L0089

European Union (2024). Regulation (EU) 2024/1991 of the European Parliament and of the council of 24 June 2024 on nature restoration and amending Regulation (EU) 2022/869. Official Journal of the European Union. 32024R1991

Fernandes, M.D.L., Quintela, A. & Alves, F.L. (2018). Identifying conservation priority areas to inform maritime spatial planning: A new approach. *Science of The Total Environment*, 639, 1088–1098. https://doi.org/10.1016/j.scitotenv.2018.05.147

Fleishman, E., Noss, R. & Noon, B. (2006). Utility and limitations of species richness metrics for conservation planning. *Ecological Indicators*, 6 (3), 543–553. https://doi.org/10.1016/j.ecolind.2005.07.005

Frazão Santos, C., Ehler, C.N., Agardy, T., Andrade, F., Orbach, M.K. & Crowder, L.B. (2019). Marine Spatial Planning. In: *World Seas: An Environmental Evaluation*. Elsevier. 571–592. https://doi.org/10.1016/B978-0-12-805052-1.00033-4

Galparsoro, I. & Borja, Á. (2021). Defining Cost-Effective Solutions in Designing Marine Protected Areas, Using Systematic Conservation Planning. *Frontiers in Marine Science*, 8, 683271. https://doi.org/10.3389/fmars.2021.683271

Gissi, E., McGowan, J., Venier, C., Carlo, D.D., Musco, F., Menegon, S., Mackelworth, P., Agardy, T. & Possingham, H. (2018). Addressing transboundary conservation challenges through marine spatial prioritization. *Conservation Biology*, 32 (5), 1107–1117. https://doi.org/10.1111/cobi.13134

Gray, C.L., Hill, S.L.L., Newbold, T., Hudson, L.N., Börger, L., Contu, S., Hoskins, A.J., Ferrier, S., Purvis, A. & Scharlemann, J.P.W. (2016). Local biodiversity is higher inside than outside terrestrial protected areas worldwide. *Nature Communications*, 7 (1), 12306. https://doi.org/10.1038/ncomms12306

Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., D'Agrosa, C., Bruno, J.F., Casey, K.S., Ebert, C., Fox, H.E., Fujita, R., Heinemann, D., Lenihan, H.S., Madin, E.M.P., Perry, M.T., Selig, E.R., Spalding, M., Steneck, R. & Watson, R. (2008). A Global Map of Human Impact on Marine Ecosystems. *Science*, 319 (5865), 948–952. https://doi.org/10.1126/science.1149345

Hammar, L., Molander, S., Pålsson, J., Schmidtbauer Crona, J., Carneiro, G., Johansson, T., Hume, D., Kågesten, G., Mattsson, D., Törnqvist, O., Zillén, L., Mattsson, M., Bergström, U., Perry, D., Caldow, C. & Andersen, J.H. (2020). Cumulative impact assessment for ecosystem-based marine spatial planning. *Science of The Total Environment*, 734, 139024. https://doi.org/10.1016/j.scitotenv.2020.139024

Hansen, G.J.A., Ban, N.C., Jones, M.L., Kaufman, L., Panes, H.M., Yasué, M. & Vincent, A.C.J. (2011). Hindsight in marine protected area selection: A comparison of ecological representation arising from opportunistic and systematic approaches. *Biological Conservation*, 144 (6), 1866–1875. https://doi.org/10.1016/j.biocon.2011.04.002

Hanson, J.O., Schuster, R., Strimas-Mackey, M., Morrell, N., Edwards, B.P.M., Arcese, P., Bennett, J.R. & Possingham, H.P. (2024). Systematic conservation prioritization with the prioritizr R package. *Conservation Biology*, e14376. https://doi.org/10.1111/cobi.14376

Huber, P.R., Greco, S.E. & Thorne, J.H. (2010). Spatial scale effects on conservation network design: trade-offs and omissions in regional versus local scale planning. *Landscape Ecology*, 25 (5), 683–695. https://doi.org/10.1007/s10980-010-9447-4

Hutchings, P. & Lunney, D. (eds) (2003). *Conserving Marine Environments: Out of sight, out of mind*. Royal Zoological Society of New South Wales. https://doi.org/10.7882/9780958608565

Ison, S., Pecl, G., Hobday, A.J., Cvitanovic, C. & Van Putten, I. (2021). Stakeholder influence and relationships inform engagement strategies in marine conservation. *Ecosystems and People*, 17 (1), 320–341. https://doi.org/10.1080/26395916.2021.1938236

Korpinen, S. & Andersen, J.H. (2016). A Global Review of Cumulative Pressure and Impact Assessments in Marine Environments. *Frontiers in Marine Science*, 3. https://doi.org/10.3389/fmars.2016.00153

Korpinen, S., Laamanen, L., Bergström, L., Nurmi, M., Andersen, J.H., Haapaniemi, J., Harvey, E.T., Murray, C.J., Peterlin, M., Kallenbach, E., Klančnik, K., Stein, U., Tunesi, L., Vaughan, D. & Reker, J. (2021). Combined effects of human pressures on Europe's marine ecosystems. *Ambio*, 50 (7), 1325–1336. https://doi.org/10.1007/s13280-020-01482-x

Kukkala, A.S. & Moilanen, A. (2013). Core concepts of spatial prioritisation in systematic conservation planning. *Biological Reviews*, 88 (2), 443–464. https://doi.org/10.1111/brv.12008

Leenhardt, P., Low, N., Pascal, N., Micheli, F. & Claudet, J. (2015). The Role of Marine Protected Areas in Providing Ecosystem Services. In: *Aquatic Functional*

Biodiversity. Elsevier. 211–239. https://doi.org/10.1016/B978-0-12-417015-5.00009-8

Linke, S., Turak, E. & Nel, J. (2011). Freshwater conservation planning: the case for systematic approaches. *Freshwater Biology*, 56 (1), 6–20. https://doi.org/10.1111/j.1365-2427.2010.02456.x

Magris, R.A., Pressey, R.L., Weeks, R. & Ban, N.C. (2014). Integrating connectivity and climate change into marine conservation planning. *Biological Conservation*, 170, 207–221. https://doi.org/10.1016/j.biocon.2013.12.032

Margules, C. & Usher, M.B. (1981). Criteria used in assessing wildlife conservation potential: A review. *Biological Conservation*, 21 (2), 79–109. https://doi.org/10.1016/0006-3207(81)90073-2

Margules, C.R. & Pressey, R.L. (2000). Systematic conservation planning. *Nature*, 405 (6783), 243–253. https://doi.org/10.1038/35012251

Marxan (2022). *Marxan Conservation Solutions*. *Marxan Conservation Solutions*. https://marxansolutions.org/

May, R.M. (1975). Islands biogeography and the design of wildlife preserves. *Nature*, 254 (5497), 177–178. https://doi.org/10.1038/254177a0

Mazor, T., Giakoumi, S., Kark, S. & Possingham, H.P. (2014). Large-scale conservation planning in a multinational marine environment: cost matters. *Ecological Applications*, 24 (5), 1115–1130. https://doi.org/10.1890/13-1249.1

McCLANAHAN, T.R., Cinner, J., Kamukuru, A.T., Abunge, C. & Ndagala, J. (2008). Management preferences, perceived benefits and conflicts among resource users and managers in the Mafia Island Marine Park, Tanzania. *Environmental Conservation*, 35 (04), 340. https://doi.org/10.1017/S0376892908005250

Mills, M., Adams, V.M., Pressey, R.L., Ban, N.C. & Jupiter, S.D. (2012). Where do national and local conservation actions meet? Simulating the expansion of ad hoc and systematic approaches to conservation into the future in Fiji. *Conservation Letters*, 5 (5), 387–398. https://doi.org/10.1111/j.1755-263X.2012.00258.x

Mills, M., Pressey, R.L., Weeks, R., Foale, S. & Ban, N.C. (2010). A mismatch of scales: challenges in planning for implementation of marine protected areas in the Coral Triangle. *Conservation Letters*, 3 (5), 291–303. https://doi.org/10.1111/j.1755-263X.2010.00134.x

Moilanen, A. (2007). Landscape Zonation, benefit functions and target-based planning: Unifying reserve selection strategies. *Biological Conservation*, 134 (4), 571–579. https://doi.org/10.1016/j.biocon.2006.09.008

Moilanen, A. (2008). Two paths to a suboptimal solution – once more about optimality in reserve selection. *Biological Conservation*, 141 (7), 1919–1923. https://doi.org/10.1016/j.biocon.2008.04.018

Moilanen, A., Franco, A.M.A., Early, R.I., Fox, R., Wintle, B. & Thomas, C.D. (2005). Prioritizing multiple-use landscapes for conservation: methods for large multi-species planning problems. *Proceedings of the Royal Society B: Biological Sciences*, 272 (1575), 1885–1891. https://doi.org/10.1098/rspb.2005.3164

Moilanen, A., Lehtinen, P., Kohonen, I., Jalkanen, J., Virtanen, E.A. & Kujala, H. (2022). Novel methods for spatial prioritization with applications in conservation, land use planning and ecological impact avoidance. *Methods in Ecology and Evolution*, 13 (5), 1062–1072. https://doi.org/10.1111/2041-210X.13819

Moshy, V.H., Bryceson, I. & Mwaipopo, R. (2015). Social-ecological Changes, Livelihoods and Resilience Among Fishing Communities in Mafia Island Marine Park, Tanzania. *Forum for Development Studies*, 42 (3), 529–553. https://doi.org/10.1080/08039410.2015.1065906

Munday, P.L., Leis, J.M., Lough, J.M., Paris, C.B., Kingsford, M.J., Berumen, M.L. & Lambrechts, J. (2009). Climate change and coral reef connectivity. *Coral Reefs*, 28 (2), 379–395. https://doi.org/10.1007/s00338-008-0461-9

Naidoo, R., Balmford, A., Ferraro, P., Polasky, S., Ricketts, T. & Rouget, M. (2006). Integrating economic costs into conservation planning. *Trends in Ecology & Evolution*, 21 (12), 681–687. https://doi.org/10.1016/j.tree.2006.10.003

Oyanedel, R., Marín, A., Castilla, J.C. & Gelcich, S. (2016). Establishing marine protected areas through bottom-up processes: insights from two contrasting initiatives in Chile. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 26 (1), 184–195. https://doi.org/10.1002/aqc.2546

Pegorelli, C., De Andres, M., García-Onetti, J., Rayo, S. & García-Sanabria, J. (2024). Marine protected areas as socio-economic systems: a method for defining socio-economic criteria in marine planning. *Frontiers in Marine Science*, 11, 1358950. https://doi.org/10.3389/fmars.2024.1358950

Pomeroy, R. & Douvere, F. (2008). The engagement of stakeholders in the marine spatial planning process. *Marine Policy*, 32 (5), 816–822. https://doi.org/10.1016/j.marpol.2008.03.017

Possingham, H.P., Wilson, K.A., Andelman, S. & Vynne, Carly.H. (2006). Protected Areas: Goals, Limitations, and Design. *Principles of Conservation Biology*, 3rd Edition, Pp. 509-533

Pressey, R.L. & Bottrill, Madeleine.C. (2008). Opportunism, Threats, and the Evolution of Systematic Conservation Planning. *Conservation Biology*, Volume 22 (No. 5), 1340–1345

Pressey, R.L., Cabeza, M., Watts, M.E., Cowling, R.M. & Wilson, K.A. (2007). Conservation planning in a changing world. *Trends in Ecology & Evolution*, 22 (11), 583–592. https://doi.org/10.1016/j.tree.2007.10.001

Pressey, R.L., Humphries, C.J., Margules, C.R., Vane-Wright, R.I. & Williams, P.H. (1993). Beyond opportunism: Key principles for systematic reserve selection. *Trends in Ecology & Evolution*, 8 (4), 124–128. https://doi.org/10.1016/0169-5347(93)90023-I

Pressey, R.L. & Tully, S.L. (1994a). The cost of ad hoc reservation: A case study in western New South Wales. *Australian Journal of Ecology*, 19, 375–384

Pressey, R.L. & Tully, S.L. (1994b). The cost of *ad hoc* reservation: A case study in western New South Wales. *Australian Journal of Ecology*, 19 (4), 375–384. https://doi.org/10.1111/j.1442-9993.1994.tb00503.x

Pressey, R.L., Watts, M.E., Barrett, T.W. & Ridges, M.J. (2009). The C-plan conservation planning system: Origins, applications, and possible futures. *Spatial*

conservation prioritization. Quantitative methods and computational tools, (United Kingdom: Oxford University Press), 211–234
Pressey, R.L., Whish, G.L., Barrett, T.W. & Watts, M.E. (2002). Effectiveness of protected areas in north-eastern New South Wales: recent trends in six measures. *Biological Conservation*, 106 (1), 57–69. https://doi.org/10.1016/S0006-3207(01)00229-4
Reimer, J.M., Devillers, R., Trouillet, B., Ban, N.C., Agardy, T. & Claudet, J. (2023). Conservation ready marine spatial planning. *Marine Policy*, 153, 105655. https://doi.org/10.1016/j.marpol.2023.105655
Reside, A.E., Butt, N. & Adams, V.M. (2018). Adapting systematic conservation planning for climate change. *Biodiversity and Conservation*, 27 (1), 1–29. https://doi.org/10.1007/s10531-017-1442-5
Rodrigues, H. & Milo-Dale, L. (2022). *Assessing the balance between nature and people in European Seas: Maritime Spatial Planning in the Baltic*. WWF.
Sacre, E., Bode, M., Weeks, R. & Pressey, R.L. (2019). The context dependence of frontier versus wilderness conservation priorities. *Conservation Letters*, 12 (3), e12632. https://doi.org/10.1111/conl.12632
Sarkar, S., Pressey, R.L., Faith, D.P., Margules, C.R., Fuller, T., Stoms, D.M., Moffett, A., Wilson, K.A., Williams, K.J., Williams, P.H. & Andelman, S. (2006). Biodiversity Conservation Planning Tools: Present Status and Challenges for the Future. *Annual Review of Environment and Resources*, 31 (1), 123–159. https://doi.org/10.1146/annurev.energy.31.042606.085844
Schultz-Zehden, A., Weig, B. & Lukic, I. (2019). Maritime Spatial Planning and the EU's Blue Growth Policy: Past, Present and Future Perspectives. In: Zaucha, J. & Gee, K. (eds) *Maritime Spatial Planning*. Springer International Publishing. 121–149. https://doi.org/10.1007/978-3-319-98696-8_6
Schuster, R., Hanson, J.O., Strimas-Mackey, M. & Bennett, J.R. (2019). Integer linear programming outperforms simulated annealing for solving conservation planning problems. https://doi.org/10.1101/847632
Serra, N., Kockel, A., Game, E.T., Grantham, H., Possingham, H.P. & McGowan, J. (2020). Marxan User Manual: For Marxan version 2.43 and above. The Nature Conservancy (TNC). chrome-extension://efaidnbmnnnibpcajpcglclefindmkaj/https://marxansolutions.org/wp-content/uploads/2021/02/Marxan-User-Manual_2021.pdf
Silvestro, D., Goria, S., Sterner, T. & Antonelli, A. (2022). Improving biodiversity protection through artificial intelligence. *Nature Sustainability*, 5 (5), 415–424. https://doi.org/10.1038/s41893-022-00851-6
Stewart, R., Noyce, T. & Possingham, H. (2003). Opportunity cost of ad hoc marine reserve design decisions: an example from South Australia. *Marine Ecology Progress Series*, 253, 25–38. https://doi.org/10.3354/meps253025
Swedish Agency Marine and Water Management (2022). Marine Spatial Plans for the Gulf of Bothnia, the Baltic Sea and the Skagerrak/Kattegat National planning in Sweden's territorial waters and exclusive economic zone. Swedish Agency Marine and Water Management.

Trouillet, B. & Jay, S. (2021). The complex relationships between marine protected areas and marine spatial planning: Towards an analytical framework. *Marine Policy*, 127, 104441. https://doi.org/10.1016/j.marpol.2021.104441

Ullah, F., Saqib, S. & Xiong, Y.-C. (2025). Integrating artificial intelligence in biodiversity conservation: bridging classical and modern approaches. *Biodiversity and Conservation*, 34 (1), 45–65. https://doi.org/10.1007/s10531-024-02977-9

United Nations (1982). United Nations Convention on the Law of the Sea. Secretary General of the United Nations. chrome-extension://efaidnbmnnnibpcajpcglclefindmkaj/https://www.un.org/depts/los/convention_agreements/texts/unclos/unclos_e.pdf

Van Der Biest, K., Meire, P., Schellekens, T., D'hondt, B., Bonte, D., Vanagt, T. & Ysebaert, T. (2020). Aligning biodiversity conservation and ecosystem services in spatial planning: Focus on ecosystem processes. *Science of The Total Environment*, 712, 136350. https://doi.org/10.1016/j.scitotenv.2019.136350

Vaughan, D. & Agardy, T. (2020). Marine protected areas and marine spatial planning – allocation of resource use and environmental protection. In: *Marine Protected Areas*. Elsevier. 13–35. https://doi.org/10.1016/B978-0-08-102698-4.00002-2

Venter, O., Magrach, A., Outram, N., Klein, C.J., Possingham, H.P., Di Marco, M. & Watson, J.E.M. (2018). Bias in protected-area location and its effects on long-term aspirations of biodiversity conventions. *Conservation Biology*, 32 (1), 127–134. https://doi.org/10.1111/cobi.12970

Vigo, M., Hermoso, V., Navarro, J., Sala-Coromina, J., Company, J.B. & Giakoumi, S. (2024). Dynamic marine spatial planning for conservation and fisheries benefits. *Fish and Fisheries*, 25 (4), 630–646. https://doi.org/10.1111/faf.12830

Virtanen, E.A., Söderholm, M. & Moilanen, A. (2022). How threats inform conservation planning—A systematic review protocol. Mingyang, L. (ed.) (Mingyang, L., ed.) *PLOS ONE*, 17 (5), e0269107. https://doi.org/10.1371/journal.pone.0269107

Virtanen, E.A., Viitasalo, M., Lappalainen, J. & Moilanen, A. (2018). Evaluation, Gap Analysis, and Potential Expansion of the Finnish Marine Protected Area Network. *Frontiers in Marine Science*, 5, 402. https://doi.org/10.3389/fmars.2018.00402

Warman, L.D., Sinclair, A.R.E., Scudder, G.G.E., Klinkenberg, B. & Pressey, R.L. (2004). Sensitivity of Systematic Reserve Selection to Decisions about Scale, Biological Data, and Targets: Case Study from Southern British Columbia. *Conservation Biology*, 18 (3), 655–666. https://doi.org/10.1111/j.1523-1739.2004.00538.x

Watson, J.E.M., Grantham, H.S., Wilson, K.A. & Possingham, H.P. (2011). Systematic Conservation Planning: Past, Present and Future. In: Ladle, R.J. & Whittaker, R.J. (eds) *Conservation Biogeography*. 1. ed. Wiley. 136–160. https://doi.org/10.1002/9781444390001.ch6

Whitehead, A.L., Kujala, H. & Wintle, B.A. (2017). Dealing with Cumulative Biodiversity Impacts in Strategic Environmental Assessment: A New Frontier for

Conservation Planning. *Conservation Letters*, 10 (2), 195–204. https://doi.org/10.1111/conl.12260

Wilson, K., Pressey, R.L., Newton, A., Burgman, M., Possingham, H. & Weston, C. (2005). Measuring and Incorporating Vulnerability into Conservation Planning. *Environmental Management*, 35 (5), 527–543. https://doi.org/10.1007/s00267-004-0095-9

Young, J.C., Jordan, A., R. Searle, K., Butler, A., S. Chapman, D., Simmons, P. & Watt, A.D. (2013). Does stakeholder involvement really benefit biodiversity conservation? *Biological Conservation*, 158, 359–370. https://doi.org/10.1016/j.biocon.2012.08.018