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REVIEW PAPER

Fluvial protected areas as a strategy to preserve riverine ecosystems—a review

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Abstract
Fluvial ecosystems are essential for life on Earth. Despite this recognition and the growing implementation of restoration programs, measures aimed at halting riverine biodiversity's decline have had limited success, so far. The implementation of protected areas has been the cornerstone of terrestrial and marine conservation. However, this strategy has only been seldomly applied to the protection of fluvial ecosystems and there is still no clear evidence of its effectiveness. We reviewed existing literature in scientific journals and reports from conservation agencies and analysed existing protection policies dedicated to rivers as well as several case studies throughout the world. Our main aim is to understand the potential advantages and drawbacks of dedicated fluvial protected areas, comparing to terrestrial protected areas and even to the total absence of protection. We also delved in the process of implementing fluvial protected areas, namely in what concerns relevant spatial scales, conservation priorities, stakeholders' involvement and mitigation measures to potential threats. In total 173 references were retained after a comprehensive search on Google Scholar, SpringerLink, Scopus and ResearchGate. These studies revealed that, despite contradictory results, terrestrial protected areas provide some degree of protection to riverine ecosystems contained within their borders, namely through increased abundances and species richness of some specific groups. Comparatively, however, dedicated fluvial protected areas, designed to accommodate the uniqueness of these systems, hold a much higher potential. Yet, data regarding its effectiveness is still scarce, mainly due to the lack of general guidelines and resources to evaluate performance following establishment, which prevents stronger conclusions.

Keywords Biodiversity · Conservation · Fluvial protected areas · Rivers

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Introduction

The environmental significance of human activities is so profound that we are presently in a distinct geological period, coined as the Anthropocene (Crutzen and Stoermer 2000) and typified by levels of global extinction several hundred times greater than expected (Olson et al. 2002; Dirzo and Raven 2003). Natural resources are progressively managed to maximize material flows to keep up with the increasing human demands, degrading the global biodiversity on which we all depend (Díaz et al. 2019). Global freshwater biodiversity has registered an average decrease of 83%, between 1970 and 2018 according to the freshwater living planet index (LPI) and based on data for 6617 monitored populations of 1398 freshwater species (WWF 2022). Riverine ecosystems, in particular, due to their topographical position and hydrological connections, are among the most endangered ones (Dudgeon et al. 2006; Reid et al. 2019). The main threats that impend over these ecosystems typically act simultaneously and include loss of natural flows and connectivity changes, water pollution, land use modifications (including of riparian corridors), overexploitation, invasion of non-native species (NNS), habitat loss and climate change (Dudgeon et al. 2006; Pittock et al. 2018; Reid et al. 2019; Feio et al. 2022a). Thus, considering rivers are habitat to an astonishing diversity of lifeforms, providing at the same time billions of people with services that include provision of clean drinking water and energy, climate regulation, navigation and nutrient cycling (Moir et al. 2016), their preservation is of utmost urgency (Darwall et al. 2011; Abell et al. 2017; Dudgeon 2019). However, most actions developed to safeguard riverine biodiversity have proved inadequate and still no global framework exists to guide policy responses in proportion with the scale and urgency of the situation (Harrison et al. 2018). Additionally, there is still a generalized bias of conservation research towards terrestrial systems (Donaldson et al. 2016; Di Marco et al. 2017).

There are several mechanisms available for the preservation of fluvial environments, including community conservation of habitats for flagship, keystone or culturally important species, formal protected areas (PA), land use planning, habitat restoration programs (UN 2018), implementation of environmental flows (e-flows), construction of fish ways and removal of obsolete dams (Reid et al. 2019).

Here, we focus specifically on the implementation of dedicated Fluvial Protected Areas (FPA).

The designation of PA has been the cornerstone of terrestrial, and more recently, marine conservation efforts (Polunin 2001). Numerous global commitments have urged for the creation of PA encompassing all major ecosystems, namely the World Commission on Environment and Development's recommendation that at least 8% of the world's terrestrial and freshwater areas be set aside in PA networks (Brundtland 1987) and the 16th Recommendation of the IVth World Parks Congress (Caracas), urging governments to ensure PA covered a minimum of 10% of each biome by the year 2000 (IUCN 1993). Also targeting for PA systems is Recommendation 19.38 of the 19th Session of the International Union for the Conservation of Nature (IUCN) General Assembly (Buenos Aires) (IUCN 1994). In 2010 the 10th Conference of Parties to the United Nations Convention on Biological Diversity (UN CBD) approved 20 Biodiversity Targets, among which was 'The conservation of at least 17% of terrestrial and inland water areas' (UN 2010). Similarly, in 2014 the IUCN issued a call for specific focus on the coverage and management of freshwater ecosystems (IUCN 2014) and in 2020, one of the World Conservation Congress resolutions (WCC-2020-Res-008) recommended that all states establish PA representative of all freshwater ecosystems (IUCN 2020). More recently, in 2022, the 15th Conference of Parties to the UN CBD adopted the Kunming-Montreal Global Biodiversity Framework, which includes the protection and conservation, through an effective system of PA and other area-based conservation measures, of at least 30% of the world's lands, inland waters, coastal areas and oceans (UN 2022). The scientific community has also been urging for increased efforts towards the establishment of dedicated PA (e.g., Moyle and Yoshiyama 1994; Li et al. 1995; Kingsford and Nevill 2005; Allan et al. 2005; Suski and Cooke 2007; Abell et al. 2008; Humphries and Winemiller 2009; Williams et al. 2011; Miranda and Pino-del-Carpio 2016; Azevedo-Santos et al. 2018; Feio et al. 2022a).

The main objectives of this study are to (i) assess the condition of riverine ecosystems within non dedicated PA, (ii) explore the existing knowledge on the benefits of FPA, (iii) look into successes and failures regarding FPA's implementation process (including several case studies) and (iv) provide recommendations towards the successful implementation and management of FPA.

Methods

An extensive literature search was conducted on the electronic databases of Google Scholar, Scopus, ResearchGate and SpringerLink to retrieve worldwide publications supporting the main goals of this study. Our screening process searched for scientific literature in English containing in the title and/or abstract, for scientific papers, and within title and/or introduction for scientific reports, the following keywords: 'freshwater ecosystems', 'riverine ecosystems', 'fluvial ecosystems', 'protected areas', 'protection areas', 'reserves', 'rivers', isolated or combined with each other. No time limits were set for the search. The adopted criteria limited the selection to studies that had been published in international journals with strict peer-review processes, in what concerns scientific papers and to renowned international conservation organizations as well as national or local organizations responsible for the management and implementation of specific FPA, in what concerns scientific reports. Regarding the case studies and considering there aren't many FPA worldwide, our search criteria limited the selection to those with reported outcomes (e.g., benefits, limitations and constraints) and representative of different tipologies of FPA. The resulting list of documents was organized into three distinct categories: (i) Fluvial ecosystems within non dedicated PA; (ii) Fluvial Protected Areas—Benefits, constraints, and case studies and (iii) FPA implementation and management. The documents with duplicated information or those that, after a detailed analysis, didn't prove relevant for the goals of the study were set aside.

Results and discussion

The bibliographic search resulted in 145 papers and 28 scientific reports, published between 1991 and 2023, with the following distribution: 35 publications on fluvial ecosystems within non dedicated PA; 43 publications on fluvial protected areas—benefits, constraints and case studies, and 95 publications on FPA implementation and management.

Fluvial ecosystems within non dedicated protected areas

Overall, the retrieved publications show that terrestrial PA provide some degree of protection towards the freshwater ecosystems contained within their limits. Considering most countries are still lacking strategic government directions aimed at the protection of fluvial ecosystems, the most common situation is their inclusion inside non dedicated PA (Tickner et al. 2020), where they are managed only as a necessary resource for preserving terrestrial biodiversity (Nel et al. 2007). Despite mixed results, most of the studies developed to assess if terrestrial PA provide some protection to fluvial ecosystems, when compared to completely unprotected areas, revealed positive effects (Fig. 1). These studies focused mainly on fish and macroinvertebrates.

These divergent results can be attributed to variables such as the legal dispositions and geographical placement of the PA analyzed and/or to different experimental designs/methodological approaches. Among the studies with positive findings are those of Baird and Flaherty 2005 (LAO PDR), Rodríguez-Olarte et al. 2006 (Venezuela), Nel et al. 2007 (South Africa), Paz et al. 2008 (Brazil), Atkore et al. 2011 (Himalaya), Abraham and Kelkar 2012 (India), Sarkar et al. 2013 (India), Gupta et al. 2015 (India), Kwik and Yeo 2015 (Singapore), Miranda and Pino-del-Carpio 2016 (Spain), Maceda-Veiga et al. 2017 (Spain), Restello et al. 2020 (Brazil) and Majdi et al. 2022 (South Africa). By opposition, the studies of Mancini et al. 2005 (Italy), Chessman 2013 (Australia), Nogueira et al. 2021 (Portugal) and Mollmann et al. 2022 (Brazil) found no protective effect towards the studied fluvial ecosystems. However, despite the positive findings, several studies have identified factors that constraint the effect of these PA in the protection of rivers (e.g., absence of whole catchment approach, limited connectivity, absence of threat control beyond PA boundaries, low surrogacy level of terrestrial taxa towards fluvial biodiversity; Darwall

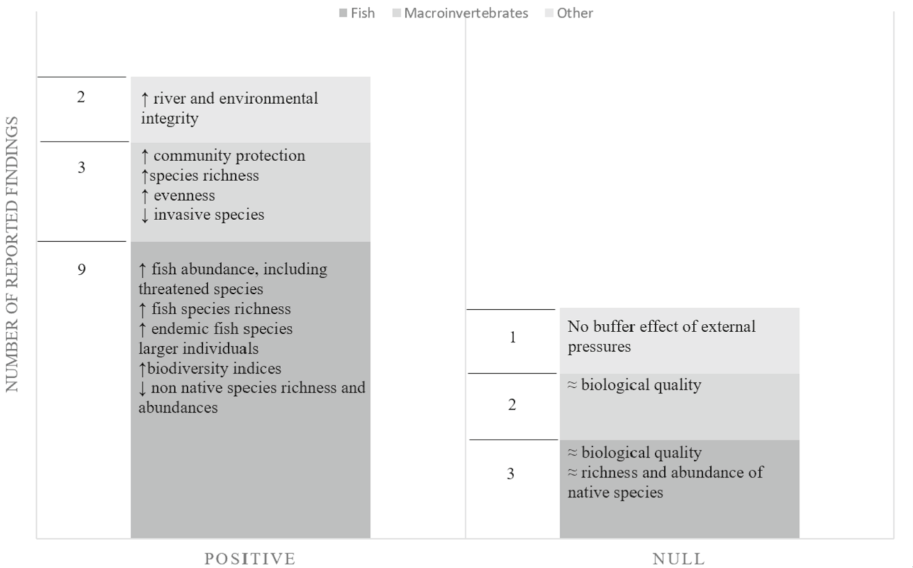

Fig. 1 Main findings resulting from studies addressing the degree of protection provided by terrestrial PA towards riverine ecosystems contained within their limits. The results are discriminated according to the element evaluated in each case (fish, macroinvertebrates and other). Some of the studies analysed simultaneously more than one element (↑≈ increase; ↓≈ decrease)

et al. 2011; Acreman et al. 2019; Leal et al. 2020; Nogueira et al. 2021). As a consequence, several authors have pointed out the limited effectiveness of these PA in what concerns the protection of riverine biodiversity (e.g., Abellán et al. 2007; Nel et al. 2007; Herbert et al. 2010; Dorji et al. 2020; Dias-Silva et al. 2021; Yousefi et al. 2022; Miyahira et al. 2023). Thus, there continues to be an urgent need to separate policies and methodologies to address PA design, designation, and management, specific to the conservation of fluvial species and ecosystems, which are too often lumped in with terrestrial habitats or marine environments (Abell et al. 2007; Saunders et al. 2002; van Rees et al. 2020).

Fluvial protected areas—benefits, constraints and case studies

Despite the ample array of policy options available within the scope of statutory protection, namely wilderness areas, national parks, river designations, community-based reserves, world heritage areas (Kingsford et al. 2016), we found that, to date the systematic implementation of fluvial reserves has been scarce (Dudgeon 2000; Saunders et al. 2002; Abell et al. 2007; Nel et al. 2009; Hermoso et al. 2016). This is justified by (i) the level of complexity of these ecosystems (Moilanen et al. 2008); (ii) the fact that many rivers and associated catchments are private, making it difficult to impose management plans (Susky and Cooke 2007); (iii) the required cooperation between multiple stakeholder groups, potentially involving multiple jurisdictions or even countries (Filipe et al. 2004); and finally, (iv) the fact that the terminology has not yet been standardized, which potentially limits the dissemination of successful examples (Susky and Cooke 2007). There are, however, exceptions (e.g., USA, Canada, Australia, South Africa, Spain, France, New Zealand, Thailand, Norway). Table 1 summarizes the ecological benefits reported, when available, as well as identified constraints associated with each case study.

Wild and scenic rivers (USA)

This was the first ever created national river conservation system. It was officially established by the Wild and Scenic River Act 1968 (WSRA) and designed to protect and enhance the free-flowing nature, water quality and outstandingly remarkable values (e.g., recreation, scenic, fish, wildlife, culture, geologic, historic and other similar values) of rivers across USA territory (Perry 2017). Rothlisberger et al. (2017) consider that the designation of a river is the riverine equivalent of creating a terrestrial PA. The Wild and Scenic River (WSR) system currently protects over 21,000 km of rivers, through 226 designations (Perry 2017, 2021), in 40 states and the Commonwealth of Puerto Rico (Moir et al. 2016). The rivers can be designated according to their wild, scenic and/or recreational values (Moir et al. 2016) and to secure their protection a Comprehensive River Management Plan is prepared for each designated river (Major et al. 2020). By securing their free-flowing nature, the designation of WSR enables natural processes to occur unimpeded (Rothlisberger et al. 2017), creating a physical template of habitats essential to native species (Thurow 2015) and contributes to an increased resilience towards climate change (Haak and Williams 2012). On the other hand, the designation usually promotes community driven projects (e.g., removal of obsolete dams, restoration of riparian buffers) aimed at the improvement and/or preservation of the overall condition of the designated river. Unfortunately, as McGarvey et al. (2021) highlight, proof that river designation itself has a measurable effect on instream biota and ecosystem health is not readily available. In fact, these authors developed one of the few studies available on the subject, focused on Virginia

Scenic Rivers Programme, similar in every respect to the WSR system. Besides the findings provided by the aforementioned study, other ecological benefits and constraints associated with this designation system were found scattered throughout different conservation agencies' reports.

Canadian heritage river system

The Canadian heritage river system (CHRS) was established in 1984 to protect Canada's outstanding rivers via a non-statutory agreement between Federal, State, and Territory governments (Nevill and Phillips 2004). It's a voluntary, community driven system that neither prohibits nor mandates activities. Each designated river has a management plan to protect its outstanding values (Nevill and Phillips 2004) and decadal monitoring reports evaluate if these values are maintained, otherwise a river may be de-designated. Rivers designated for natural values, which represent the majority, may not be impounded or have impoundments outside their boundaries (Canadian Heritage Rivers Board 2002). There are 41 designated rivers spanning over 11,000 km across Canada (Nevill and Phillips 2004). Similarly to the WSR (USA), data attesting the benefits provided by the designation towards aquatic communities are scarce.

Kakadu national park (Australia)

Kakadu National Park (KNP), in northern Australia, was declared between 1979 and 1991 to protect the entire catchment of a large floodplain river (Parks Australia 2016). The park encompasses almost all of the catchment of the South Alligator River, the entire catchment of the smaller West Alligator, and significant parts of the Wildman River and East Alligator River catchments (Adams et al. 2015). KNP is listed as a Wetland of International Importance under the Ramsar Convention and was also inscribed in the World Heritage List in 1981, under both cultural and natural criteria (BMT WBM 2010). The park covers an area of 19,816 km^2 and comprises a highly dynamic and productive fluvial component, connected all the way from headwaters to mouth (Parks Australia 2016). Following declaration, potentially pernicious activities were restricted, and a management plan was established (Adams et al. 2015), providing a framework for decision-making and resource allocation (Parks Australia 2016). However, according to Parr et al. (2009), this plan lacks precision and measurable targets, making it hard to assess the extent of management success in achieving biodiversity objectives. In fact, the management's impact has never been consistently evaluated (Adams et al. 2015) and there has been no long term, systematic monitoring of threatened plant or animal species, with the exception of two species of marine turtles (Parr et al. 2009).

Community-based fish reserves (Thailand)

To counteract the effects of over-fishing, several southeast Asia local communities (e.g., Philippines, Cambodia, Thailand, India), have designated hundreds of no-fishing riverine reserves (Loury and Ainsley 2020). These reserves are usually surrounded by intensely fished areas (McCann et al. 2016). Thus, to secure a tight surveillance and to minimize travel distances, the reserves' boundaries usually coincide with the most upstream and downstream homes in the village, and penalties, whether monetary or non-monetary, are applied to the transgressors (Koning et al. 2020). Koning et al. (2020) assessed the effectiveness of one of those reserves, within the Salween River basin (northern Thailand), testing the fish communities' response. These authors found that despite their small size (0.2 to 2.2 ha), these grassroot reserves benefited fish populations, which shows that empowering communities to manage local resources can have better results than top-down management (Koning et al. 2020).

Blanice river nature reserve (Czechia)

In 1989, the Blanice River nature reserve (63 km^2), in Czechia, was designated to protect the upper river network of the Blanice River against anthropogenic eutrophication and to restore habitat conditions for sensitive riverine species, in particular the freshwater pearl mussel (*Margaritifera margaritifera*) (Fraindová et al. 2022). Between 1990 and 1992, eradication of mineral fertilization, liming and pesticide application, as well as land use transition from croplands to traditional grasslands, produced a semi-natural landscape dominated by meadows and forests (Staponites et al. 2022). Staponites et al. (2022) assessed the impact of management practices adopted after designation on the water quality, comparing data before 1992 and after 2000, and found measurable improvements in surface water quality.

National freshwater ecosystem priority areas (South Africa)

Another example, more of a priorization tool than an attribution of protected status, is the South African National Freshwater Ecosystem Priority Areas (NFEPA) project (2008–2011), that came into place due to the high levels of threat reported for this country's freshwaters (Nel et al. 2011b). Its aim was to identify freshwater ecosystem priority areas (FEPA) using a range of criteria dealing with the maintenance of key ecological processes and conservation of ecosystem types and species associated with rivers, wetlands and estuaries (Driver et al. 2011). Another goal of this project was to develop a basis to enable effective implementation of protection measures towards the identified FEPAs, including free flowing rivers. Guidelines were defined for each of the four different management units identified (wetland FEPAs, river FEPAs, sub-quaternary catchments associated with river FEPAs and upstream management areas), linking specific land use practices and activities to changes in water quality, changes in water quantity and changes in habitat and biota (Driver et al. 2011). The identified FEPA can be used both in proactive planning processes (e.g., catchment-wide planning processes, water use regulation, land use regulation) and in reactive decision-making processes (Driver et al. 2011).

The publications selected in our study indicated also that, although FPA's main goal is conservation, they also contribute to an ample array of ecosystem services (e.g., support to essential natural processes, maintenance of freshwater fish populations and provision of other food sources, groundwater recharge, provision of raw materials, water purification, research and education, recreation, and disaster risk reduction; Balmford et al. 2015; Dudley et al. 2018; Feio et al. 2022b). In addition, FPA contribute to protect terrestrial species depending on riverine ecosystems (e.g., refuges in degraded landscapes or at times of drought, corridors for migration and dispersal; van Rees et al. 2020). In fact, as emphasized by Hermoso et al. (2012), FPA offer interesting opportunities for terrestrial conservation and vice versa. Therefore, targeting both realms simultaneously is recommended, generating higher benefits than those achieved by individual terrestrial and fluvial PA (Alvarez-Romero et al. 2011; Hermoso et al. 2012).

Table 1 Summary of reported FPA's positive impacts on riverine ecosystems and constraints associated to each of the case studies (↑≈ increase)

FPA	Positive impacts	Constraints
WSR (USA)	↑ Fish habitat suitability; protection of instream flows (McGarvey et al. 2021) ↑ Water quality; successful control of NNS (SuAsCo-WSR Stewardship Council 2018) ↑ Populations of resident and/or nesting important bird species; removal of obsolete dams (WRA-IPA-UD 2016) Complex aquatic and riparian habitats (Rothlisberger et al. 2017)	Activities upstream or downstream of designated segments are not regulated; ↑ land cover changes, pollution and contamination due to recreational activities; instream flows poorly protected; fish and wildlife populations managed by state agencies (Rothlisberger et al. 2017) Private land not regulated by public land managers; prone to conflicts among stakeholders (Keith et al. 2008) and among central and provincial authorities (Blauwkamp and Longo 2002)
CHRS (Canada)	↑ natural area cover; ↑ reproductive success of key species (Essex Region Conservation Authority 2021) Fluvial ecosystems rated with good condition and with an improving trend (Parks Canada 2011)	Possibility of de-designation if the nominating province chooses to do so; prone to conflicts among central and provincial authorities (Blauwkamp and Longo 2002)
KNP (Australia)	Prevention of 2.7% increase in cover of NNS (*Mimosa pigra*) (Adams et al. 2015) Persistence of most native species (Parr et al. 2009) Riverine component with very good condition, comparing with other wetlands globally (Parks Australia 2016)	Potential opposing interests between indigenous landowners' interests and biodiversity preservation (Parr et al. 2009)
Blanice river nature reserve (Czechia)	Measurable improvements in surface water quality (Staponites et al. 2022)	–
Community-based fish reserves (Thailand)	↑ fish species richness, density and biomass; ↑ local fisheries due to spillover effect towards adjacent unprotected areas (Koning et al. 2020)	Metapopulations persistence potentially compromised due to isolation of sub-populations; only partial protection for long-distance migrants (Koning et al. 2020) Reduced impact on other threats affecting overall effectiveness (Loury and Ainsley 2020)

Table 1 (continued)

FPA	Positive impacts	Constraints
NFEPA (South Africa)	Stabilization of local populations of endangered fish (O'Brien et al. 2014) 7 years after FEPA identification, no major deterioration in ecological condition and connectivity has been detected (Petersen et al. 2022)	Not enough relevance has been given to flows (Paxton et al. 2016) Declaration of Nature Reserves depends on the landowner's willingness and on the balance between conservation and economic needs (Kotzé 2012) Requires more investment on research (Nel et al. 2011b)

Fluvial protected areas implementation and management

Our search highlighted the most relevant factors for the successful implementation and management of FPA refered in publications, namely the definition of conservation priorities, stakeholder's engagement and minimization of potential threats (e.g., land use disturbances, flow alterations, introduction of NNS and climate changes).

Defining conservation priorities

The first step in the implementation of any kind of PA is the definition of priorities (Margules et al. 2002; Hermoso et al. 2016). We identified two major approaches to systematic reserve design, based either on a scoring procedure or on the concept of complementarity. The former ranks sites, independently from each other, in order of priority according to defined criteria (e.g., species richness, rarity or vulnerability) (Boon 2000; Abellán et al. 2005). On the other hand, methods based on complementarity, such as systematic conservation planning (SCP), are more recent and have received growing interest (e.g., Margules and Pressey 2000; Margules and Sarkar 2007; Hermoso et al. 2012) by providing, in a context of limited options and resources, a quantitative method for conservation prioritization using observed or modelled surrogate distributions (Pressey et al. 1993). SCP can be defined as "*a structured stepwise approach to mapping conservation area networks, with feedback, revision and reiteration, where needed, at any stage*" (Margules and Sarkar 2007). According to Frank and Sarkar (2010), the SCP frameworks usually include several sequential steps (Fig. 2).

Nevill and Phillips (2004) advocate that these sequential steps could be used to form the basis of a strategy aimed at establishing representative FPA in a given planning region (PR). In fact, we identified several studies that have already used this approach to put forward proposals for the development of FPA, namely in Australia (e.g., Linke et al. 2007) and France (e.g., Decker et al. 2017).

Although most SCP approaches have been originally developed within terrestrial and marine realms, several adaptations for riverine systems, including methods to incorporate longitudinal, lateral, vertical, and temporal connectivities, have been made (Moilanen et al. 2008; Hermoso et al. 2012).

The results arising from this priority analysis are aimed to inform the management and decision-making processes governing environmental conservation (Wilson et al. 2009).

Once established, PA have the potential, if correctly managed, to provide long-term safety to associated biota, maintaining natural processes and viable populations, while minimizing potential threats (Margules and Pressey 2000; Margules et al. 2002). The regular performance assessment of management responses through biophysical, governance and socio-economic indicators (Pomeroy et al. 2005) should be a standard part of PA planning and funding (Adams et al. 2015). For instance, Xie et al. (2019) detected that FPA established in specific sections of Yangtze river (China) were performing poorly towards the protection of targeted fish species, which indicated the need for further optimizations. There is no doubt about FPA's potential to effectively protect populations and habitats, however, the impacts of its management actions have rarely been evaluated (Adams et al. 2015), as demonstrated by some of the case studies aforementioned. Important shortcomings are the limited data availability (Juffe-Bignoli et al. 2014), limited resources, high levels of natural variability and spatio-temporal connectivity (Adams et al. 2015) and also

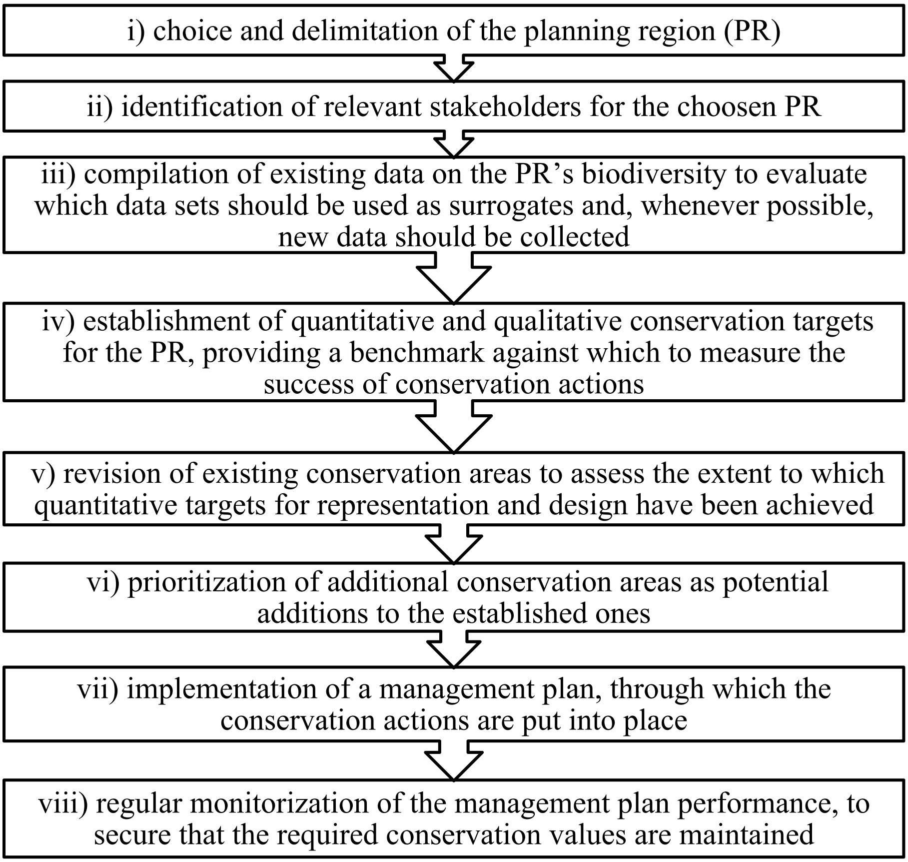

Fig. 2 Sequential steps included in SCP systematic conservation planning frameworks, according to Frank and Sarkar (2010)

to the lack of general guidelines for assessing FPA following establishment (Loury and Ainsley 2020). The IUCN Greenlist initiative was developed, among other things, to bridge this gap, functioning as a global certification programme, promoting PA that are effectively managed and that demonstrate measurable conservation outcomes (IUCN and WCPA 2017). This strategy can also be applied to FPA.

Public and stakeholder intervention

Our search on the engagement of stakeholders in the protection of rivers highlighted the need to involve all sectors in an integrated way, to avoid colliding interests (Darwall et al. 2014) and prevent devastating effects. Indeed, in the past, the establishment of nature reserves and national parks led to the displacement of many indigenous populations justified with the preservation of wilderness, namely in the USA and South Africa (Spence 1999). Nonetheless, this paradigm has changed, and presently the way of life and the aspirations of resident populations are increasingly considered (Dudley and Parish 2006). To reach a generalized consensus, conservation planners need to implement a stakeholder involvement framework based on a flexible, transparent, participative approach and also on principles of prior informed consent and shared responsibilities, from the onset of the conservation planning exercise (Linke et al. 2011; Borrini-Feyerabend et al. 2013; Mauerhofer et al. 2015). As Darwall et al. (2014) point out, unless all sectors and stakeholders work together in an integrated way, the interests of one sector will inevitably collide with those of another. Besides public participation in decision making, several authors (e.g., Garcia-Llorente et al. 2016; Feio et al. 2022b) underscore that it's equally important to endorse the values of nature and biodiversity across all sectors of society, increasing ecosystem knowledge, improving social cohesion, potentiating nature preservation, and advocating the vital services provided by healthy ecosystems.

Minimizing threats within fluvial protected areas

Different strategies have been presented to eliminate or minimize deleterious impacts, not only within FPA, but also in terrestrial PA fortuitously encompassing fluvial ecosystems, from which it's still possible to derive benefits, according to various studies analysed (e.g., Nel et al. 2007; Herbert et al. 2010). In fact, authors like Dudley and Stolton (2008), Nel et al. (2009), Thieme et al. (2012) and Abell et al. (2017) have put forward a few changes to terrestrial PA that, adding to the minimizing actions addressed below, would improve their preservation benefits for fluvial ecosystems. Some of these changes include (i) restoring longitudinal and lateral connectivity, (ii) avoiding the use of rivers as boundaries, (iii) avoiding the development of visitor infrastructures near priority fluvial ecosystems, (iv) managing aquatic recreational activities, (v) restoring riparian buffers, (vi) managing processes and threats internal and external to the boundaries of the PA, (vii) expanding the existing PA to encompass unprotected headwaters, natural large-scale catchment processes, biodiversity hotspots, functional processes and connectivity pathways, (viii) assuring that the PA management plans address and monitor, regularly, riverine conservation objectives and (ix) prioritizing resources for those PA that include large portions of catchments and/or that contain populations of rare, endemic or threatened species. Acreman et al. (2019) strongly recommends that more attention is given to such opportunities, especially where the implementation of dedicated FPA is unlikely or financially unfeasible. This strategy also has the additional advantage of being more politically acceptable, more cost-effective, and less disruptive to human communities than the creation of a completely new FPA (Pringle 2017).

Below we discuss the solutions found in literature regarding how FPA should be implemented and managed to deal with, and reduce the effect of, land use and flow alterations, introduction of NNS and climate change.

Land use alterations's Any conservation planning must acknowledge, not only that FPA will need to be part of larger protected landscapes (Nevill and Phillips 2004), but also the interconnections between multiple stressors and between the different riverine habitats within the drainage basin (Pittock et al. 2018). Hence, several authors (e.g., Saunders et al. 2002; Dudgeon et al. 2006; Linke et al. 2007, 2008; Hermoso et al. 2013; Adams et al. 2015; Mollmann et al. 2022) consider that the ideal solution would be to select management units encompassing all relevant connections (e.g., catchments or sub-catchments). According to these authors, the integration of target fluvial areas into the basin's management would secure habitat patchiness, connectivity pathways and associated ecological processes. Additional benefits include (i) increased cooperation across multiple agency jurisdictions, (ii) integration of public and private land management efforts, (iii) increased comprehensiveness regarding NNS control, (iv) increased climate change resilience, (v) promotion of environmentally sensitive compatible uses (Williams et al. 2011) and (vi) higher integration between fluvial and terrestrial conservation initiatives (Klein et al. 2009). However, gaining management control over large areas of land is complex, challenging and, in most cases, unfeasible (Dudgeon et al. 2006; Hermoso et al. 2016). To overcome this constraint, there are alternative approaches compatible with the underlying principle of managing entire catchments as a whole. One of those approaches involves the establishment of hierarchical frameworks (e.g., Abell et al. 2007), which contributes not only to fulfill spatial needs, but also to ensure effective protection in a more flexible way (Saunders et al. 2002). Accordingly, there is always a central core with a given fluvial feature requiring protection, surrounded by a series of sequential buffer zones, in which some human activities are allowed at varying intensities (Abell et al. 2007). Only low-impact activities are allowed in areas closer to the central core, while potentially more harmful practices are excluded or relegated to more distant areas (Saunders et al. 2002). This strategy is increasingly being accepted as an appropriate fluvial conservation framework (e.g., Linke et al. 2011; Nel et al. 2011a), not only because many biodiversity patterns, habitats and processes are shaped by a spatial and temporal hierarchy (Poff 1997; Margules and Pressey 2000), but also for being relatively easy to integrate with local community-based conservation initiatives (Saunders et al. 2002). Hermoso et al. (2016), for example, have used this method to refine conservation recommendations in the Iberian Peninsula, reducing the overall area needed to expand Natura 2000 network, in order to adequately cover fluvial conservation targets in that region.

Flow alterations Natural flows create channel features, maintain floodplain connectivity, enable groundwater recharge, mitigate flooding (Tickner et al. 2020) and support productive fisheries and flood recession agriculture, which sustain millions of people (Moir et al. 2016). Hence, and considering riverine species have evolved life strategies in direct response to natural flows and water regimes, any man-made changes, namely through artificial in stream structures (e.g., dams, weirs, culverts, levees), lead to biodiversity losses (Bunn and Arthington 2002). These structures fragment fluvial ecosystems and disrupt movements of water, species, sediments, and nutrients (Grill et al. 2019; Rivaes et al. 2022), resulting in decreases in native species' abundances and diversity, accompanied by proliferation of NNS (Maceda-Veiga et al. 2017). Fish are particularly affected, since their free movement for reproduction, feeding and habitat colonization is blocked, leading to potential genetic impoverishment (Hermoso et al. 2012; Branco et al. 2014). In fact, WWF's LPI 2020 reports a 93% average decline of migratory freshwater fish in Europe since 1973 (WWF 2020). Despite these negative implications, the atribution of protection status doesn't necessarily shield riverine ecosystems from this threat. In the Mediterranean region alone, around 2000 hydropower projects are already proposed within the boundaries of National Parks, Biosphere Reserves and other PA (Freyhof et al. 2020). At the global scale, there are around 1,200 dams within PA and more than 500 are under construction or planned for the next two decades (Kroner et al. 2019). In view of this scenario, active management through e-flows (Arthington et al. 2018), dam discharges, inter-basin transfers and groundwater extractions, gain an even bigger relevance, since they can reproduce natural discharge patterns (Poff 1997). The removal of artificial and obsolete in-stream structures is also gaining momentum and being increasingly considered a viable option for a sustainable watershed management (Bernhardt et al.

2005). The EU Biodiversity Strategy aims to return at least 25,000 km of rivers into a free-flowing status by 2030, by removing obsolete barriers (EC 2022), of which more than 7000 have already been removed (Mouchlianitis 2023). Fish passage devices also contribute to increase fluvial connectivity, allowing fish to move freely (Branco et al. 2014). Hydrologic connectivity must be carefully managed, within and beyond FPA's boundaries (Pringle 2001) and any effort to manage and/or restore it should be directed to where they are most effective (Saunders et al. 2002). Considering more than 90% of a river's flow derives from headwaters, the main actions, especially when resources are limited, should focus on these areas (Imbert and Stanford 1996).

Introduction of non-native species (NNS) Biological invasions in rivers have been increasing, especially in the last 60 years (Nunes et al. 2015) and are expected to become even more frequent, due to the increased human pressure on natural habitats, climate change and expansion of international trade (Genovesi and Monaco 2014; Zamora-Marín et al. 2023).

Nowadays, NNS can be introduced either purposely (e.g., agriculture, aquaculture, pet trade), or accidentally (e.g., ballast water, bait buckets, fouling or concealment in transported goods) (Mack 2003). Impacts include changes to ecosystem services and processes, changes in biodiversity (e.g., behavioural shifts, restructured food webs) (Carbonell et al. 2017), increased biological homogenization (Kortz and Magurran 2019) and negative effects on local economies, man made structures and livelihoods (Shackleton et al. 2014). Within PA, NNS' richness patterns are usually linked to the surrounding environment's characteristics, such as road networks, land use and human population density, as well as PA's surface area and protection status (Foxcroft et al. 2011; Gallardo et al. 2017; Moustakas et al. 2018). Accordingly, Holenstein et al. (2021) and Genovesi and Monaco (2014) strongly recommend that the focus of NNS management be extended beyond PA boundaries. The best way to address NNS requires updated and detailed information in order to prevent their establishment, regulating activities associated with their introduction and promoting dispersion barriers (Saunders et al. 2002; Zamora-Marín et al. 2023). These barriers, preferably of natural origin, can include the establishment of buffer zones around PA (Foxcroft et al. 2011) or, for instance, the maintenance of natural flows, which continue to allow endemic species' migration, simultaneously preventing NNS invasion (Moyle and Yoshiyama 1994; Marchetti and Moyle 2001). A good example of effective prevention efforts is the 'Check, clean, dry: didymo controls' programme of the Fiordland National Park (New Zealand), aimed at preventing the establishment of the invasive freshwater diatom (microalgae) *Didymosphenia geminata* (didymo), by encouraging visitors to check, clean and dry all gear before leaving the lake edge and moving into lake tributaries or other waterways (Biosecurity New Zealand 2023).

When NNS are already present, but not fully established, complete eradication is preferable and more cost-effective than long-term control (Zavaleta et al. 2001), although it should only be carried out where ecologically feasible and when there is financial and political support to be effectively completed (IUCN 2000). When NNS are established, an active, focused and adaptative management is necessary, not only to reduce their negative effects, but also to prevent their further dispersion (Saunders et al. 2002). Eradication and containment measures vary according to the target species and respective life cycles, but can include, among others, traps, baits, mechanical cleaning actions, selective removal, establishment of physical barriers, chemical treatments and biological control (Anastácio et al. 2019). Wood-Pawcatuck Watershed Association, for example, conducted a 3 year project to control purple loosestrife (*Lythrum salicaria*), an abundant NNS on the Pawcatuck River, using *Galerucella* spp. beetles, which resulted in a progressive decrease of *L. salicaria* and in increased abundance of native species (e.g., *Eupatorium perfoliatum*) (WPWA 2013).

Climate change Although PA play an important role in species and ecosystem conservation, Finlayson and Pittock (2018) consider that managing them in the same way, focusing on static reference conditions, for decades to come, may have adverse consequences, not only but also because climate change induces alterations in the natural ecosystems. As Mascia and Pailler (2011) point out, PA may fail if they no longer contain the elements they were designed to protect. Climate change will continue to increase the pressure on rivers through alterations in flow patterns and intermittency, modifications in the frequency, magnitude and timing of droughts or floods, as well as changes to water quality and biological communities (Palmer et al. 2008). Our review revealed two publications (Finlayson et al. 2017; Biggs et al. 2012) that provided a set of general principles to help guide PA management within the context of climate change and that could also be adopted for FPA. These principles include (i) definition of objectives that accommodate and compensate for climate change, (ii) establishment of flexible governance and adaptive co-management frameworks, (iii) selection of easily reversed, no-regret adaptations, (iv) implementation of long-term management strategies that identify triggers for new actions, (v) development of monitoring campaigns to evaluate management strategies, (vi) promotion of diverse, redundant and well connected systems and (vii) management of slow variables. Engaging partners and knowledge holders, supporting staff progress, committing to ongoing knowledge development, and thinking at broader scales are also relevant actions to consider (Gross et al. 2016). Within established FPA there are several adaptations that can improve their resilience to climate change, namely (i) the integration into the protected perimeter of parts of the fluvial landscape that may be more resilient to climate change and/or which can provide refugia and ecological complexity (Turak et al. 2011); (ii) the conservation of essential areas to secure the viability and resilience of the FPA (e.g., riparian zones, upstream headwaters) (Major et al. 2020) and (iii) the protection of larger and environmentally heterogeneous areas (Heino et al. 2009). Additionally, on FPA encompassing storage dams able to discharge the right amount of water, at the right temperature, e-flows' management can also be an important intervention (Olden and Naiman 2010). On the other hand, free flowing rivers should become conservation priorities (Pittock and Finlayson 2011), since they increase the resilience of these ecosystems by providing open pathways for species movement to suitable habitats, in response to climate change (Palmer et al. 2008; Groves et al. 2012).

Conclusions

Through a comprehensive literature review of 173 research papers and other publications, we have compared the condition of riverine ecosystems within terrestrial PA and in dedicated FPA, exploring the advantages and constraints of the latter. Despite the mixed results, it's safe to say that terrestrial PA confer some degree of protection to riverine ecosystems contained within their boundaries. However, to rely on the protection provided by these PA alone, is inadequate and insufficient, considering the increasing amount of threats acting upon freshwater ecosystems. Moreso many of the reviewed cases reported significant representation gaps in terrestrial PA towards riverine biodiversity. In comparison, dedicated FPA have a higher potential, as long as adequately managed, to provide enhanced protection, considering they are specifically designed for these ecosystems. Accordingly, we have explored, through our literature review, some of the most relevant factors for the successful implementation and management of these dedicated PA. However, attribution of statutory protection represents the starting point and only dedicated focus on monitoring aquatic ecosystem's integrity, using reliable indicators, will secure an effective management that meets the intended objectives. Oddly, our literature review resulted in a much higher number of studies addressing the effectiveness of non dedicated terrestrial PA towards riverine ecosystems, than studies addressing the effectiveness of dedicated FPA towards the ecosystems they were designed to protect. This discrepancy has already been reported by other authors (e.g., Loury and Ainsley 2020) and can be due to the fact that terrestrial PA are much more abundant and widespread than FPA and also to the general lack of data, resources and guidelines. Besides this gap, there are other challenges and hindrances to overcome, namely in what concerns governance issues, requiring as Vance-Borland et al. (2008) highlight, the development of common purpose, policy integration, and strategy alignment across different disciplinary backgrounds, overlapping mandates, and potentially conflicting policy contexts. Nonetheless, if correctly managed, the long-term benefits clearly outweigh the short-term costs and obstacles.

FPA cannot, however, be the sole strategy for tackling biodiversity loss in fluvial ecosystems. Instead, it should be used in complementarity with other strategies, in order to maximize its outcomes and effectively contribute to the preservation of riverine ecosystems for both present and future generations.

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Data availability Data sharing not applicable to this article as no datasets were generated or analysed during the current study.

Declarations

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