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Nature-based solutions for natural hazards and climate change

Edited by
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Published in
Frontiers in Environmental Science
Frontiers in Earth Science
Frontiers in Ecology and Evolution
Frontiers in Marine Science

ISSN 1664-8714
ISBN 978-2-83251-158-9
DOI 10.3389/978-2-83251-158-9

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Nature-based solutions for natural hazards and climate change

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Citation
Reguero, B. G., Renaud, F., Di Sabatino, S., Jongman, B., van Zanten, B., Beck, M. W., Cohen-Shacham, E., eds. (2023). *Nature-based solutions for natural hazards and climate change*. Lausanne: Frontiers Media SA. doi: 10.3389/978-2-83251-158-9

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frontiers | Frontiers in Environmental Science

TYPE Editorial
PUBLISHED 13 December 2022
DOI 10.3389/fenvs.2022.1101919

Editorial: Nature-based solutions for natural hazards and climate change

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OPEN ACCESS

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SPECIALTY SECTION
This article was submitted to Interdisciplinary Climate Studies, a section of the journal Frontiers in Environmental Science

RECEIVED 18 November 2022
ACCEPTED 24 November 2022
PUBLISHED 13 December 2022

CITATION
Reguero BG, Renaud FG, Van Zanten B, Cohen-Shacham E, Beck MW, Di Sabatino S and Jongman B (2022), Editorial: Nature-based solutions for natural hazards and climate change. *Front. Environ. Sci.* 10:1101919. doi: 10.3389/fenvs.2022.1101919

KEYWORDS

nature-based solutions, climate change, natural hazards, disaster risk, risk management, nature-based adaptation, ecosystem-based adaptation

Editorial on the Research Topic
Nature-based solutions for natural hazards and climate change

Manuscript contribution to the field *

The interactions between the coupled systems of climate, ecosystems and human society increasingly represent the basis of emerging risks. Nature-based Solutions (NbS) are increasingly seen in this context as a fundamental approach to address natural hazards and climate risks. Despite the global momentum for NbS in climate and environmental agendas, their implementation and uptake face policy, institutional, technical and financial challenges. This multidisciplinary collection demonstrates the increasing scientific evidence on the effectiveness of NbS for natural hazards and climate risks, which is essential to increase acceptance and uptake. The examples, case studies and experiences presented across watersheds, agricultural lands, and coastlines, urban and rural settings, offer new knowledge to address key challenges and demonstrate the potential of NbS to align climate, environmental and sustainable development goals. The collection aims to help advance the use of NbS in multiple contexts, but especially in regions at the forefront of climate change and natural hazard risks.

Why a focus on nature-based solutions for natural hazards and climate change?

Climate change, environmental degradation, and disasters from natural hazards are some of the most pressing global threats society faces today. The interactions between the coupled systems of climate, ecosystems (including biodiversity) and human society increasingly represent the basis of emerging risks (IPCC 2022) (Figure 1). This collection on "Nature-based Solutions for natural hazards and climate change" exemplifies such needed integration of knowledge across the natural, ecological, social and economic sciences.

From 2000 to 2019, 7,348 major recorded disasters claimed 1.23 million lives and affected 4.2 billion people resulting in approximately US$2.97 rillion in global economic losses (UNDRR 2020). This represents a sharp increase over the previous two decades, which is explained by a rise in climate-related disasters, including extreme weather events. The economic cost from climate-related events, caused by atmospheric-driven phenomena, totaled $329 billion in 2021 and marked the third-highest loss on record after adjusting for inflation, only behind the years 2017 and 2005 (AON 2021). Human-induced climate change is also affecting extreme events and causing widespread impacts to people and nature, beyond natural climate variability (IPCC 2022). In developing countries and areas most exposed to climate change, climate impacts exacerbate existing vulnerability and injustices, undermining sustainable development efforts (IPCC 2022).

Nature-based Solutions (NbS) are increasingly seen as a fundamental approach to address these challenges and an essential component to achieve the goals of the Paris Agreement on Climate Change (UNFCCC 2015) and of the Sendai Framework for Disaster Risk Reduction (UNISDR 2015). There are multiple definitions of NbS. However, the Fifth session of the United Nations Environment Assembly of the United Nations Environment Programme adopted a multilaterally agreed definition as (UNEP 2022): "*actions to protect, conserve, restore, sustainably use and manage natural or modified terrestrial, freshwater, coastal and marine ecosystems, which address social, economic and environmental challenges effectively and adaptively, while simultaneously providing human well-being, ecosystem services and resilience and biodiversity benefits*". Therefore, NbS can be considered central to Climate Resilient Development, as conceptual and operational framework to deal with climate risks, adaptation, and mitigation efforts, while also benefiting the environment and human well-being. NbS also underpin the Sustainable Development Goals, by enhancing the provision of vital ecosystem services and job creation (e.g., Edwards et al., 2013).

Despite the increasing momentum for NbS across global climate and biodiversity agendas, implementation remains rather limited (IPCC 2022). While investments in NbS for natural hazards and climate change are increasing, both in emerging and advanced economies, much more is needed for NbS to effectively complement gray infrastructure for climate adaptation (UNEP 2021; UNFCCC 2022; World Bank 2022). Barriers to bringing NbS investment to scale include policy, institutional, technical and financial challenges. This Research Topic provides a collection of evidence, new findings and insights that contribute to address some of these challenges to advance NbS for climate resilience across NbS types and environments, including coastlines, forests, watersheds, agriculture and small islands. The different contributions are summarized below.

Summary of contributions

The articles in this collection provide multidisciplinary insights and scientific evidence on the effectiveness of NbS for reducing impacts from natural hazards and climate risks; recommendations for the planning and design of NbS; and case studies on their benefits.

As editors, we highlight six particularly compelling conclusions from the contributions that are essential to increase acceptance and uptake of NbS are:

1) It is possible to identify the conditions under which NbS can effectively deliver critical risk reduction benefits;
2) The knowledge base on the effectiveness of NbS is rapidly increasing;
3) Addressing stakeholder acceptance and perceptions are critical to increase the uptake and scale of NbS projects, and collaborative co-design and participatory approaches can help;
4) Large-scale experiences with NbS in watersheds for flood mitigation have been proved to deliver environmental benefits.
5) Lack of economic information on benefits and costs remains a key barrier to broader uptake of NbS.

The specific contributions are summarized below:

1. It is possible to identify the conditions under which NbS can effectively deliver critical risk reduction benefits;

Roelvink et al. uses physics-based simulations to investigate where coral reef restoration projects could be most effectively implemented for reducing coastal flooding. The study compares different types of reefs morphology and offers important guidance for reef-based coastal protection demonstrating that the flood reduction effectiveness of a project can vary significantly depending on the reef profile types, location and dimensions. As a result, coastlines fronted by three-slope profiles are relatively unprotected from wave action in unrestored conditions, but could benefit the most from reef restoration projects.

Van Bijsterveldt et al. provides insight for optimizing mangrove restoration, one of the most widely used NbS globally, by combining planting with ecological restoration, which relies on natural mangrove regeneration by facilitating trapping seeds in target ponds. However, seaward mangrove expansion through planting alone, without additional measures to restore mangrove habitat, are likely to be unsuccessful and represent poor practices (or similarly, planting non-pioneer species at newly colonized sites).

In many cases, NbS will be combined with engineered or "grey" measures in hybrid approaches. One good example is the use of vegetation to add safety to engineered structures. Schoutens et al. provide the first direct experimental proof of the stability of marshes during breaching scenarios and high flow velocities showing that they could mitigate water flow during discharges after a dike is breached. Historic analysis from flood disasters in North Europe have shown that saltmarshes reduce the chance and size of the breaching of engineered defenses by reducing water flow (Zhu et al., 2020). The article by Schoutens et al. provides new important insight for using marshes in multi-layer defense systems and dike strengthening strategies, with potential application in many countries.

2. The knowledge base on the effectiveness of NbS is rapidly increasing.

Moraes et al. provides a review of implemented cases and projects in coastal and estuarine areas of Europe, to capitalize on lessons learnt and support future implementation. The results show an increasing number of experiences between 2005 and 2015, but dominated by hybrid designs and restoration projects, mostly for wetlands, while the creation of new habitat represents

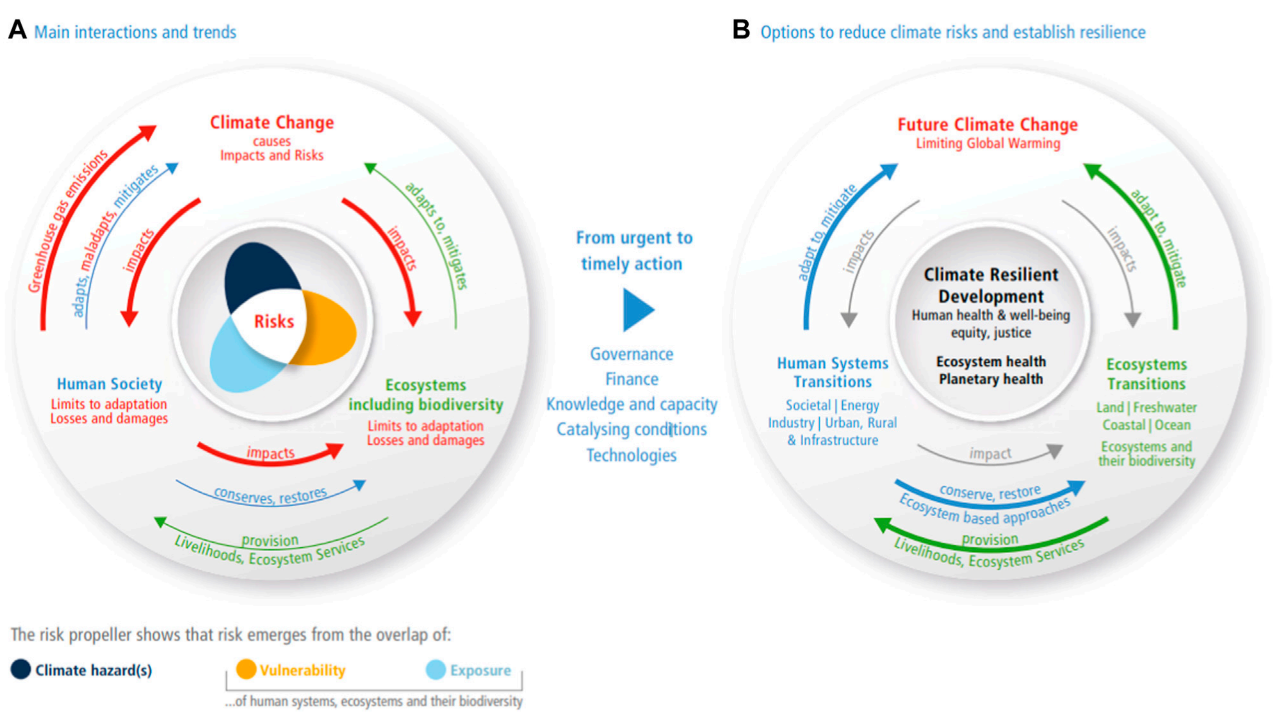

FIGURE 1
From climate risk to climate resilient development: climate, ecosystems (including biodiversity) and human society as coupled systems. Interactions among the coupled systems climate, ecosystems (including their biodiversity) and human society are the basis of emerging risks from climate change, ecosystem degradation and biodiversity loss and, at the same time, offer opportunities for the future. **(A)** Human society causes climate change. Climate change, through hazards, exposure and vulnerability generates impacts and risks that can surpass limits to adaptation and result in losses and damages. Human society can adapt to, maladapt and mitigate climate change, ecosystems can adapt and mitigate within limits. Ecosystems and their biodiversity provision livelihoods and ecosystem services. Human society impacts ecosystems and can restore and conserve them. **(B)** Meeting the objectives of climate resilient development thereby supporting human, ecosystem and planetary health, as well as human well-being, requires society and ecosystems to move over (transition) to a more resilient state. The recognition of climate risks can strengthen adaptation and mitigation actions and transitions that reduce risks. Taking action is enabled by governance, finance, knowledge and capacity building, technology and catalysing conditions. Transformation entails system transitions strengthening the resilience of ecosystems and society (Section E in IPCC, 2022). In **(A)** arrow colours represent principle human society interactions (blue), ecosystem (including biodiversity) interactions (green) and the impacts of climate change and human activities, including losses and damages, under continued climate change (red). In **(B)** arrow colours represent human system interactions (blue), ecosystem (including biodiversity) interactions (green) and reduced impacts from climate change and human activities (grey). Taken from: IPCC, 2022, Figure TS.2, page 42, and chapter 1.2, Figure 1.2.

only 20% of the projects. Most of the projects were supported by more than one funding source, which highlights the importance of co-financing, although they overwhelmingly relied on public funding sources. The analysis also reveals a lack of reporting of monitoring and co-benefits, which are often used to promote the project but that remain, in most cases, largely unquantified. The review may be timely, given that 37% of the European Recovery and Resilience Facility (about 267 billion euros) could support climate investments and reforms that may include coastal NbS for adaptation and mitigation benefits (European Commission 2021).

Kiddle et al. presents examples of nature-based approaches to adapt and mitigate the impacts of climate change and urbanization in Pacific islands, which are on the frontline and amongst the most vulnerable to the impacts of climate change (IPCC, 2022). Based on the analyses of experiences in three Pacific Island Nations, Kiddle et al. highlights the critical role of traditional ecological knowledge in shaping localized, place-based, nature-based adaptation. Solutions like "ridge to reef" approaches, restoration and protection of coastal vegetation and watersheds, or the intensification of home gardens and urban greening, are increasingly available for Small Island Developing States.

Smith et al. further contributes to the effectiveness evidence base by reviewing NbS experiences in Bangladesh for the mitigation of climate impacts and natural hazards and their contribution to sustainable development. Bangladesh is one of the most climate vulnerable countries in the world, where climate risks are compounded by environmental degradation and socio-economic challenges. Smith et al. finds robust evidence that, across landscapes, well-designed NbS can be effective in reducing hazard risks, adapting to climate change and reducing greenhouse gas emissions, while empowering marginalized groups, reducing poverty, supporting local economies and enhancing biodiversity. Furthermore, four enabling factors can maximize benefits: policy support; participatory approaches; strong and transparent governance; and finance and land tenure.

NbS can also be effective against landslides and erosion events, through interventions that reinforce slopes with vegetation. Gonzalez-Ollauri et al. propose a comprehensive set of key performance indicators towards building a more robust evidence base on NbS performance for landslide and erosion prevention. The proposed framework aims to address a gap in demonstrating the multifunctional performance of nature-based landslide prevention and mitigation, by combining indicators and metrics that balance monitoring, engineering performance, and the provision of ecosystem functions and services.

Yet, gaps in knowledge across landscapes remain. Simelton et al. find limited evidence for and underutilization of NbS in agricultural systems, especially in developing countries. The authors propose a framework that establishes four essential functions to add functionality, purpose and scale when designing NbS in agriculture projects, which aims to overcome the divide between production- and conservation-oriented approaches. Key challenges involve economic valuations; social aspects, like farmers' willingness to adopt new practices; and policy dimensions, which should address governance barriers.

3. Addressing stakeholder acceptance and perceptions are critical to increase the uptake and scale of NbS projects, and collaborative co-design and participatory approaches can help.

Lupp et al. discuss the implementation of NbS through participative approaches, and studies stakeholder perceptions of NbS in rural mountain areas, which have been less attended by research compared to urban contexts. Despite the importance of NbS in the political and research agendas, they find limited knowledge at the on-the-ground level. In rural mountain areas, many landowners (in particular farmers) initially perceive NbS as a limitation to economic outcomes of their land, in contrast with urban areas, where public landowners or real estate developers may be more attracted by the creation of multiple co-benefits. Despite these challenges, upscaling and replication of good NbS interventions were perceived to be an attractive opportunity. As a solution, they recommend creating economic and business cases based on real-life examples.

Anderson et al. examine public acceptance of NbS, by exploring interactions between societal attitudes and values towards risk, nature, and place. The authors use surveys from three distinct sites where specific hazards are addressed through NbS: landslides and coastal erosion; eutrophication and algal blooms; and river flooding and water scarcity. Their findings confirm demand for evidence of effectiveness of NbS to counter initial skepticism, hesitant attitudes and cautious positive perceptions. To increase public acceptance, they recommend framing NbS in relation to place-based values, historic characteristics, and evidence of the effectiveness of projects for risk reduction.

Similarly, O'Donnell et al. investigate resident perceptions of the performance of mangroves, beaches and hardened shorelines after Hurricane Irma in the lower Florida Keys. Their study indicates a disconnect between perceptions and performance outcomes: although mangroves cost less to repair (averaging $64.33 USD per meter) than hardened shorelines ($105.14 USD per meter) and were perceived as the most effective for storm protection, the majority of Florida Keys residents own hardened shorelines. Beaches were perceived as the most damaged shoreline type, followed by mangroves, and hardened shorelines. The study provides important timely guidance in the aftermath of hurricane Ian (2022) in the Caribbean and Florida, but it is also useful in long-term strategies given the increasing impacts of tropical cyclones globally.

4. Large-scale experiences with NbS in watersheds for flood mitigation have been proved to deliver environmental benefits.

Gooden and Pritzlaff discuss "Dryland Watershed Restoration with Rock Detention Structures" as NbS to address land degradation, mitigate drought, watershed erosion and flooding, and contribute to revegetation. In the arid southwestern United States and northern Mexico, rock detention structures (RDS) are a technology adapted from traditional indigenous practices that include a variety of types, such as check dams, one rock dams, and gabions. RDS can represent simple, cost effective, hand-built solutions, with proven positive impacts on stream flow, reduction of peak runoff, and increased sedimentation. They also indicate benefits for increased biodiversity and wildlife abundance, increased in vegetative cover; and surface water provisioning over time. However, five barriers to replication and scalability include: limited awareness of degradation and benefits of restoration; lack of legislation, policies, and regulation; technical capacity; finance; and research on costs and carbon sequestration potential.

Norman provides commentary on the potential of RDS for land restoration. Norman describes RDS scalability throughout landscapes, perseverance over time, and contributions to a restoration stewardship economy that supports RDS. In particular, the commentary elaborates on the scalability in space and time of RDS and how they differ from green infrastructure in built environments, which is implemented to passively harvest rainwater. Instead of retaining water, RDS are designed for allowing water to slowly pass through, infiltrate the soils and regenerate landscapes. The commentary also provides more information on their costs and benefits, one of the critical gaps.

Kasada et al. examine the influence on flood hazard and biodiversity of land-use pattern changes in rural Japan. The study demonstrates that land-use change can reduce flood risk while also positively influencing species richness and abundance. The demonstrated benefits in local biodiversity of targeted management of an agricultural landscape (dominated by paddy fields) fill a gap in the lack of quantitative evaluations of the impacts of ecosystem-based disaster risk reduction on biodiversity.

5. Lack of economic information on benefits and costs remains a key barrier to broader uptake of NbS.

To guide investments in NbS, stated preference studies have become a common tool to evaluate the benefits of NbS in developing countries. Hagedoorn et al. provide a comparison of time and money payment methods for evaluating the willingness to pay in Ghana. In "money payments", respondents make trade-offs between changes in ecosystem services and monetary compensation. Time payments, however, serve as an alternative where the willingness to pay is also a function of non-monetary contributions (e.g., time). The authors suggest that a combination of wage-based and non-wage-based conversion approaches is the most adequate approach to convert time to money valuations.

The examples, case studies and experiences presented across watersheds, agricultural lands, and coastlines in both urban and rural settings, add new knowledge to address key challenges of NbS and further demonstrate their potential to align climate, environmental and sustainable development goals. Overall this collection helps advance our understanding of the scalability and uptake of NbS in multiple contexts, but especially in regions at the forefront of climate change and natural hazard risks.

Author contributions

BGR wrote the first version of this editorial. All the editors contributed to the final version of this Editorial.

Acknowledgments

All the editors also thank the reviewers for their service, insight, and comments during the revisions of this Research Topic. BR acknowledges the support from an Early-Career Research Fellowship from the Gulf Research Program of the National Academies of Sciences, Engineering, and Medicine for the support provided during the guest editorial service that enable this Research Topic, although the content is solely responsibility of the authors and does not necessarily represent the official views of the Gulf Research Program of the National Academy.

Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Publisher's note

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frontiers in Marine Science

ORIGINAL RESEARCH
published: 10 May 2021
doi: 10.3389/fmars.2021.653945

Coral Reef Restorations Can Be Optimized to Reduce Coastal Flooding Hazards

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Coral reefs are effective natural coastal flood barriers that protect adjacent communities. Coral degradation compromises the coastal protection value of reefs while also reducing their other ecosystem services, making them a target for restoration. Here we provide a physics-based evaluation of how coral restoration can reduce coastal flooding for various types of reefs. Wave-driven flooding reduction is greatest for broader, shallower restorations on the upper fore reef and between the middle of the reef flat and the shoreline than for deeper locations on the fore reef or at the reef crest. These results indicate that to increase the coastal hazard risk reduction potential of reef restoration, more physically robust species of coral need to be outplanted to shallower, more energetic locations than more fragile, faster-growing species primarily being grown in coral nurseries. The optimization and quantification of coral reef restoration efforts to reduce coastal flooding may open hazard risk reduction funding for conservation purposes.

Keywords: coral reefs, wave runup, ecosystem services, coastal protection, reef degradation, reef restoration, climate change, coastal risk

OPEN ACCESS

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Specialty section:
This article was submitted to
Ocean Solutions,
a section of the journal
Frontiers in Marine Science

Received: *15 January 2021*
Accepted: *01 April 2021*
Published: *10 May 2021*

Citation:
Roelvink FE, Storlazzi CD, van Dongeren AR and Pearson SG (2021) Coral Reef Restorations Can Be Optimized to Reduce Coastal Flooding Hazards. Front. Mar. Sci. 8:653945. doi: 10.3389/fmars.2021.653945

INTRODUCTION

Coral reefs not only help sustain the economy of 500 million people in tropical coastal communities (Hoegh-Guldberg et al., 2019), but also protect them from wave-driven flooding and coastal erosion (Elliff and Silva, 2017; Reguero et al., 2018), especially in the face of climate change (Storlazzi et al., 2018). Coral reefs can substantially reduce coastal flooding by efficiently attenuating ocean wave energy (Ferrario et al., 2014). Over coral reefs, wave height rapidly decays by wave breaking, resulting in a mean surface elevation increase (setup) (Vetter et al., 2010). Waves are also damped by bottom friction caused by corals. At the shoreline, residual wave energy drives wave runup, which together with setup, results in coastal flooding that is greatest onshore of steep, narrow, and hydraulically smooth reefs (Quataert et al., 2015). The hydrodynamic behavior of coral reefs is generally well characterized by coastal engineering models (Van Dongeren et al., 2013; Taebi and Pattiaratchi, 2014; Quataert et al., 2015). Utilizing such models, the role of coral reefs in coastal hazard risk reduction has recently been assessed (Reguero et al., 2019; Storlazzi et al., 2019) and quantified to be in the order of $billions annually in the United States alone.

Coral degradation due to climate-change and anthropogenic activities has caused the loss of reef elevation (Yates et al., 2017) and roughness, thereby increasing coastal flooding hazards (Quataert et al., 2015). Coral reefs are under threat from both local stressors such as land-based pollution and overfishing (Carson et al., 2019) and global stressors such as global warming and ocean acidification (Pandolfi et al., 2011). The increasing frequency and magnitude of these impacts have retarded coral reefs' natural ability to recover, contributing to an annual decline in coral cover of 1–2% (Bruno and Selig, 2007). To reverse this trend, coral restoration is increasingly being undertaken (Bostrom-Einarsson et al., 2020), and research efforts to support such measures have primarily focused on growth and outplanting techniques (Chan et al., 2018), monitoring (Montoya-Maya et al., 2016), upscaling of restoration works (Levy et al., 2018), long-term ecological resilience (Van Oppen et al., 2015), and reef management (Rinkevich, 2014; Lirman and Schopmeyer, 2016).

Coral reef restoration is suggested to reduce the flood risk to, and increase the resiliency of, tropical coastal communities (e.g., Ferrario et al., 2014; Beck and Lange, 2016). Ferrario et al. (2014) evaluated the wave attenuation capacity of entire coral reef features (including reef crest, reef flat), but noted both ecological and engineering (e.g., restoration location, height, and roughness) knowledge gaps in designing reef restoration projects for hazard mitigation. The current study tackles these knowledge gaps by providing practical guidelines to maximize restoration efforts for coastal risk reduction, as the large spatial scales of coral reefs and operational constraints of recovery efforts necessitate an efficient approach in designing and restoring coral reefs. We utilized a physics-based, hydrodynamic model to evaluate how the height, width, and relative location of a restoration on various reef morphologies found in nature influences the wave-driven runup reduction potential of the restoration to provide information on their hydrodynamic performance and to guide restoration design for coastal hazard risk reduction. Such information will significantly increase the efficiency of coral restoration efforts, assisting a range of stakeholders in their efforts on not only coral reef conservation and management, but also coastal hazard risk reduction, and possibly open new financing options for reef restoration *via* pre-disaster hazard mitigation funds or post-disaster restoration funds.

MATERIALS AND METHODS

The extent to which coral restoration can enhance the protection of tropical coastal communities was investigated with a calibrated and validated physics-based hydrodynamic model that propagates offshore wave conditions over a one-dimensional reef profile with and without coral restorations to predict the wave-induced runup that leads to coastal flooding. First, coral reef profiles observed in nature were classified with a statistical tool into four dominant reef profile types. For each representative reef profile shape, coral reef restorations were designed based on operational and environmental constrictions. The effect of the different restorations on wave driven flooding was assessed with the XBeach model (Roelvink et al., 2009), using a wide variety of potential coral restoration configurations (different widths, heights, and locations), reef morphologies and hydrodynamic forcing conditions (water depths, wave heights, and wave periods).

Classification of Coral Reef Profiles

A global dataset of more than 30,000 coral reef profiles, encompassing a wide array of coral reef shapes and topographic features from the Pacific and Atlantic Oceans, was classified to obtain representative coral reef profile shapes. The dataset, compiled by Storlazzi et al. (2019), consists of measurements of the depth at cross-shore intervals of 2 m, starting 20 m above mean water level and extending to 30 m below mean water level. Several geographic regions are captured in the dataset: Hawaii, Florida, Guam, Puerto Rico, United States Virgin Islands, American Samoa, and the Northern Mariana Islands. Fringing reefs, barrier reefs, and atoll reefs are accounted for in the dataset.

Following Costa et al. (2016), geometrically distinct representative reef profiles were created by clustering with the k-means algorithm based on the similarity between all cross-shore depth locations. First, profiles were divided into length bins to prevent the focusing of cluster centers (here the mean reef profile of all profiles that belong to a cluster or group) on longer profiles. Length bin borders were derived from visual inspection of initial clustering on the full dataset: [0–600; 600–1,500; 1,500–2,600; 2,600–4,000; 4,000–17,000 m]. For each length bin, 10–15 cluster centers were obtained to find a good balance between bathymetric variability and compactness of the results. The number of clusters is chosen using the Davies–Bouldin evaluation index (Davies and Bouldin, 1979), which indicates how well the reef profiles fit to the cluster and not to other clusters. In total, 61 cluster centers, hence 61 clustered profile groups, were obtained with the k-means algorithm. However, as cluster centers are the mean of all profiles in a clustered profile group and hence not necessarily actual reef profiles, representative, observed reef profiles from the dataset were extracted for each clustered profile group. For each profile group, five representative reef profiles were extracted from the dataset to capture the bathymetric profile variability within each clustered group as well. All representative profiles, 305 in total, were then visually categorized based on their reef shape, from which 10 shapes were discerned (see **Supplementary Figure 1**), with an approximated frequency of occurrence as noted in **Supplementary Table 1**.

Four profiles with distinct hydrodynamic behavior and relatively high frequency of occurrence were extracted for subsequent modeling steps (**Figure 1**), representing approximately 70% of the analyzed data. The four types are categorized as a "fringing" reef (18% occurrence), a "convex" reef (in which the transition between the fore reef and reef flat is gradual, 11% occurrence), a "linear" reef (31% occurrence), and a "three-slope" reef (a steep nearshore slope, followed by a shelf, and an offshore fore reef slope, 10% occurrence). The barrier reef (13% occurrence) was omitted from the analysis for two reasons. First, wind-wave growth dominates the wave field inshore of barrier reefs and thus nearshore waves that cause flooding are relatively independent of barrier reef height (Gallop et al., 2014; Drost et al., 2019) and thus coral reef restorations far from the shore (more than a kilometer) on the barrier reef will have a minimal impact on coastal hazard risk reduction. Furthermore, the 1D modeling approach used here is not valid for wide (e.g., barrier) reef systems where wave refraction

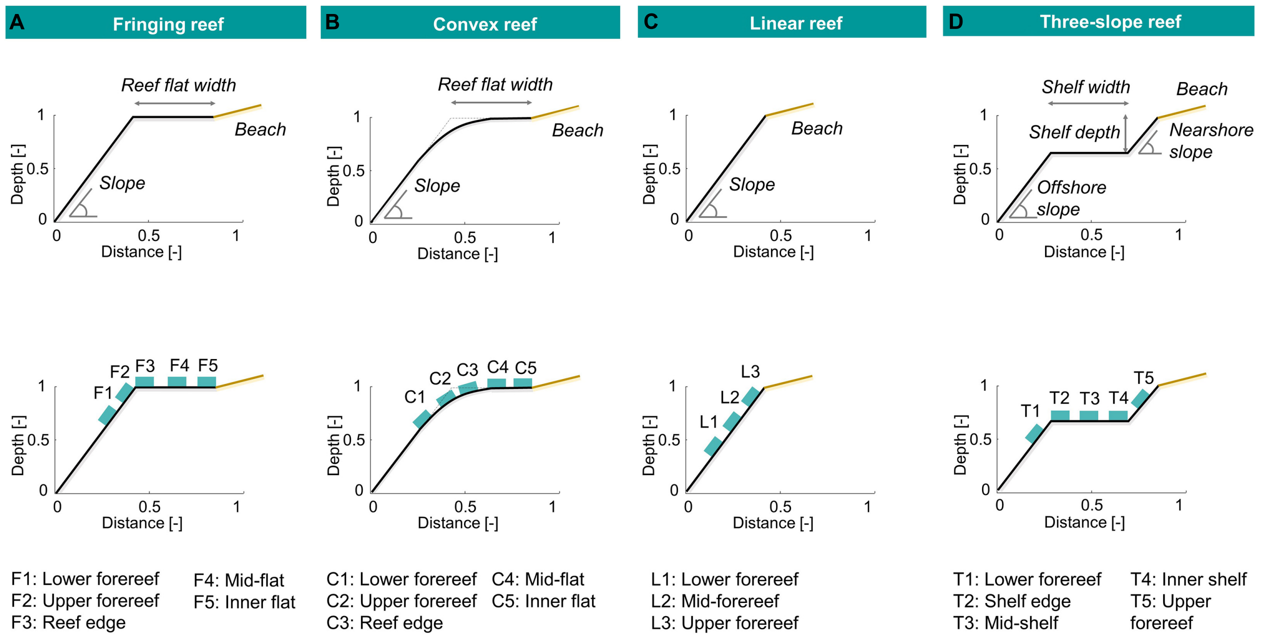

FIGURE 1 | Overview of the four characteristic reef profile types and restoration locations. **(A)** Fringing, **(B)** convex, **(C)** linear, and **(D)** three-slope reef profile types.

and horizontal circulation patterns are important for waves and wave-driven water levels (Lowe et al., 2010), with outflow through the channels in the barrier reef generally offsetting wave-driven set-up at the reef crest.

For each profile shape, different profile parameter values (reef flat/shelf width and slope) were used in the modeling study to capture some of the profile variability encountered in nature: fringing reef: reef flat width = (100 and 250 m), fore reef slope = (0.1 and 0.5), convex reef: reef flat width = (100 and 250 m), fore reef slope = 0.1, linear reef: fore reef slope = (0.025, 0.1, and 0.5), three-slope reef: shelf width = (100 and 250 m), and offshore and nearshore slope = (0.1 and 0.5).

Restoration Schematization

Restorations were schematized according to Storlazzi et al. (2021, in press), who worked with stakeholders, scientists, and decision-makers to develop generalized restoration scenarios that considered (i) likelihood of delivering flood reduction benefits, (ii) existing coral restoration practices, and (iii) permitting factors such as depth for potential navigational hazards. Restorations can be either purely "green," entailing solely outplanting corals, or "gray-green hybrid," entailing emplacement of structures (such as ReefBalls) and then outplanting corals on top of them (Shaver and Silliman, 2017; Bostrom-Einarsson et al., 2020).

Here, restorations with dimensions of 5–25 m width and 0.25–1.25 m height were placed at 18 locations (F1–5, C1–5, L1–3, and T1–5 for the fringing reef, convex reef, linear reef, and three-slope reefs, respectively) across the four reef profile shapes (**Figure 1**). For each reef profile type, the different restoration configurations were designed based on restrictions of coral restoration works (restoration depth, width, and roughness). Restoration depth boundaries of 2 m (lower limit) and 7 m (upper limit) were set by operational constraints: deep enough to not interfere with small vessel traffic, and shallow enough to make outplanting by divers feasible. The effects of reef restorations were investigated for three different restoration widths: 5, 10, and 25 m, based on proposed reef restoration efforts by federal agencies and non-governmental organizations; these widths were constrained by projected costs of restoration measures per unit length of shoreline. Restorations were modeled as an increase in bed level: 0.25 m for "green" restoration representing outplanting 0.25-m high corals from a nursery and 1.25 m for "gray-green" hybrid restoration representing outplanting corals from a nursery on top of a 1-m high artificial structure (such as a ReefBall) across the width of the restoration, with enhanced hydrodynamic roughness due to the presence of the outplanted corals. Friction values were based on a meta-analysis of reef wave breaking studies by Storlazzi et al. (2019), who proposed a friction value (c_f) of 0.15 for 90–100% coral cover and 0.01 for no coral cover. These values were adopted for the restored section and the unrestored reef bathymetry, respectively.

Wave-Driven Flood Modeling

XBeach (XB) is a physics-based nearshore model that solves the horizontal equations for flow, wave propagation, sediment transport, and changes in bathymetry (Roelvink et al., 2009). The non-hydrostatic version of XBeach (XB-NH, De Ridder et al., 2021), which solves wave evolution of both short and infragravity waves allows for an accurate estimation of all coastal runup components (Lashley et al., 2018). In particular, the XB-NH+ version was used, a reduced two-layer model which improves the dispersive behavior of the model compared to

XB-NH. The XB-NH model was extensively validated for sandy coastlines (Roelvink et al., 2018), vegetated coasts (Van Rooijen et al., 2015), and most importantly, coral reef environments (Quataert et al., 2015, 2020; Lashley et al., 2018; Storlazzi et al., 2018).

Here, one-dimensional XB-NH+ models were set up based on the schematized reef bathymetric profiles; the grid resolution varies with depth, with the grid being coarser at greater depths and farther offshore than in the nearshore. A semi-infinite beach extended up to +30 m above mean sea level so that runup could be directly evaluated across the entire range of hydrodynamic forcing. The offshore boundary location was set by depth restrictions of $n < 0.75$ (the ratio of wave group speed over phase speed, which influences the generation of LF waves at the boundary), kh (relative depth, with k the wavenumber, and h depth), and for the wave height to depth ratio to prevent the breaking of waves at the boundary. XB-NH+ parameter values were obtained from a calibration study using field observations on coral-reef lined coasts (Quataert et al., 2020). Although Quataert et al. (2020) calibrated the XBeach model by varying the reef roughness, here the reef roughness is parametrized following Storlazzi et al. (2019), using friction values (c_f) of 0.15 for the reef restoration, 0.01 for the (degraded) unrestored reef bathymetry, and 0.001 for the sandy beach.

The XB-NH+ models were forced with a range of water levels and wave conditions commonly observed in nature (Kolijn, 2014; Quataert et al., 2015; Shope et al., 2016). Kolijn (2014) provided a variation in significant wave heights of measured storm events from 1.0 to 4.5 m, whereas Quataert et al. (2015) and Cheriton et al. (2016) measured maximum wave heights of 6 m. Hence, wave heights of 2, 4, and 6 m were investigated in this study. These wave heights were combined with wave steepness values of 0.01 (typical for a swell event) and 0.05 (typical for a storm event). Kolijn (2014) also found that reef water levels of 68 investigated reef sites mostly range between 0 and +2.5 m relative to the reef flat, with tidal excursions of 0.5–1.0 m. Hence, water levels of 0.5, 1.0, 2.0, 3.0, and 4.0 m were imposed, where those of 0.5 and 1.0 m were excluded for simulations of reef flat restoration across the fringing and convex reef to prevent drying of the restoration. The corresponding JONSWAP spectra were imposed at the offshore boundary. Following Lashley et al. (2018), at cross-shore locations indicated in **Figures 2–5**, wave height components were discerned based on spectral analysis of local water level time series, using a split frequency of $0.5^{*}f_p$ (peak frequency of incident SS waves) to distinguish the SS waves (frequency $> 0.5^{*}f_p$) from the LF waves (frequency $< 0.5^{*}f_p$).

Water level time series were separated into incoming and outgoing components *via* the method of Guza et al. (1984) from local water level elevations and current velocities. Runup, defined as the elevation water reaches up the beach slope with a 2% exceedance value, was extracted from the runup water level time series, by solving for the SS wave, LF wave, and setup components

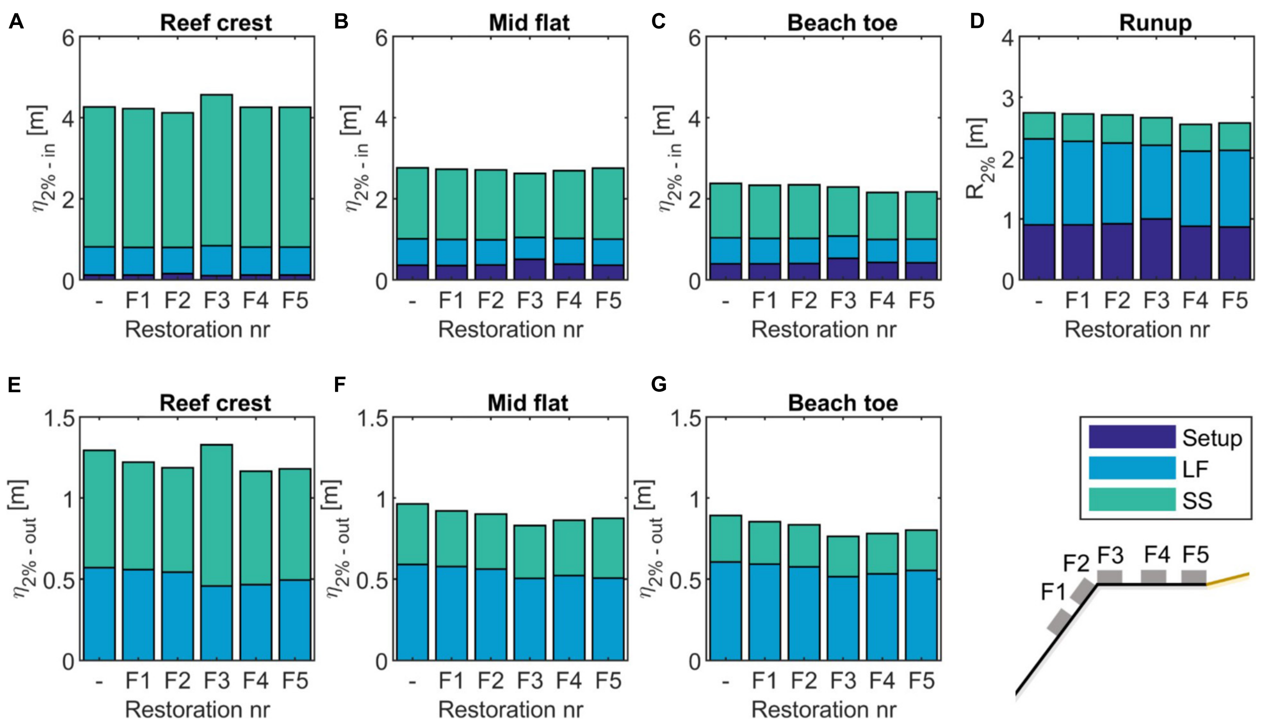

FIGURE 2 | The influence of different restoration configurations on water levels across a fringing reef profile. The mean short (SS) wave, long (LF) wave, and setup water level extremes at three locations across the reef: **(A)** Reef crest, **(B)** mid reef flat, and **(C)** beach toe for profiles with no restoration [Restoration nr (-)] and restored profiles (Restoration nr = F1 to F5, corresponding to **Figure 1**), averaged across model variations. **(D)** Mean runup heights. Water levels are separated into incoming **(A–C)** and reflected **(E–G)** water levels.

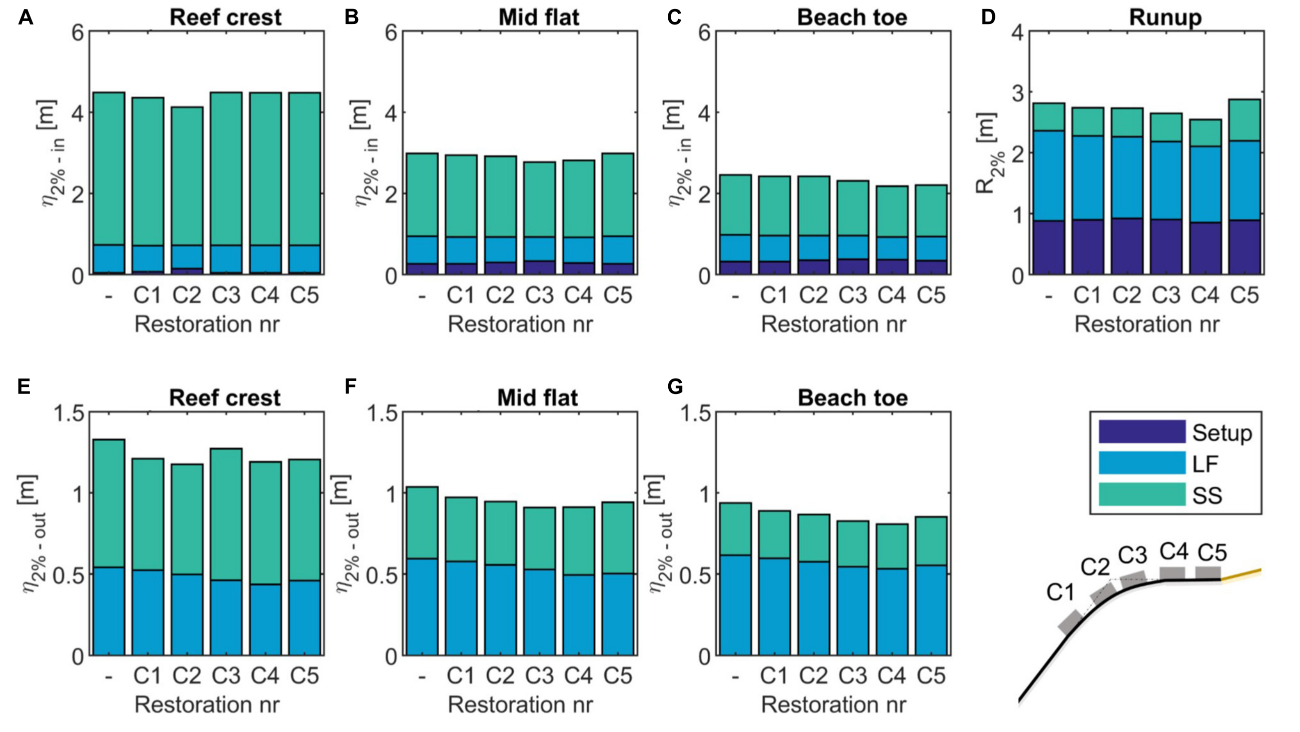

FIGURE 3 | The influence of different restoration configurations on water levels across a convex reef profile. The mean short (SS) wave, long (LF) wave, and setup water level extremes at three locations across the reef: **(A)** Reef crest, **(B)** mid reef flat, and **(C)** beach toe for profiles with no restoration [Restoration nr (-)] and restored profiles (Restoration nr = C1 to C5, corresponding to **Figure 1**), averaged across model variations. **(D)** Mean runup heights. Water levels are separated into incoming **(A–C)** and reflected **(E–G)** water levels.

of the runup. The steady setup component was obtained by extracting the mean water level relative to the still water level. The SS wave and LF wave runup components were obtained from the detrended water level time series by spectral composition. The total water level results were sorted in ascending order to select the 2% exceedance value.

RESULTS

To identify promising coral reef restoration strategies for protecting the adjacent shoreline, runup reduction by coral reef restorations was investigated. Wave transformation across unrestored and restored representative reef profiles was investigated to identify processes governing the runup at the shoreline.

Wave Transformation Across Unrestored Coral Reef Profiles

To isolate the hydrodynamic effects of a restoration, we first evaluated wave transformation across unrestored coral reef profiles with the one-dimensional XBeach models. Results (not shown here) confirm general patterns found in previous studies (e.g., Cheriton et al., 2016; Pearson et al., 2017; Buckley et al., 2018). A narrow surf zone (steep slope) promotes the quick dissipation of sea-swell ("SS," 5–25 s periods) waves and the generation of wave-induced setup and breakpoint-forced low-frequency ("LF," 25–1,000 s periods) motions, resulting in increased wave runup and thus coastal flooding potential on steeper-sloped coasts. Increased water depth, narrower reef width and/or lower roughness reduce frictional dissipation across the reef (Quataert et al., 2015). Large reflection values at the beach, combined with greater water depths and lower roughness, promotes the amplification of LF wave heights by resonance and thus increases coastal flood risk (Cheriton et al., 2016).

Wave Transformation Across Restored Coral Reef Profiles

Coral reef restorations are expected to affect the wave transformation process and the subsequent wave-driven runup and potential coastal flooding by modifying the bathymetry and seabed roughness. Various restored reef profiles were tested for a range of hydrodynamic forcing conditions in order to account for the many non-linear interactions between reef morphology and hydrodynamics (Quataert et al., 2015).

Wave transformation is characterized by the water levels across the reef $\eta_{2\%}$ and the runup $R_{2\%}$, defined as the upper 2% of the water level time series at a specified location (e.g., Merrifield et al., 2014; Pearson et al., 2017; **Figures 2–5**). Wave reflection and dissipation across a coral restoration are the main processes affecting wave transformation relative to unrestored profiles, as observed across all four profile types. The seaward

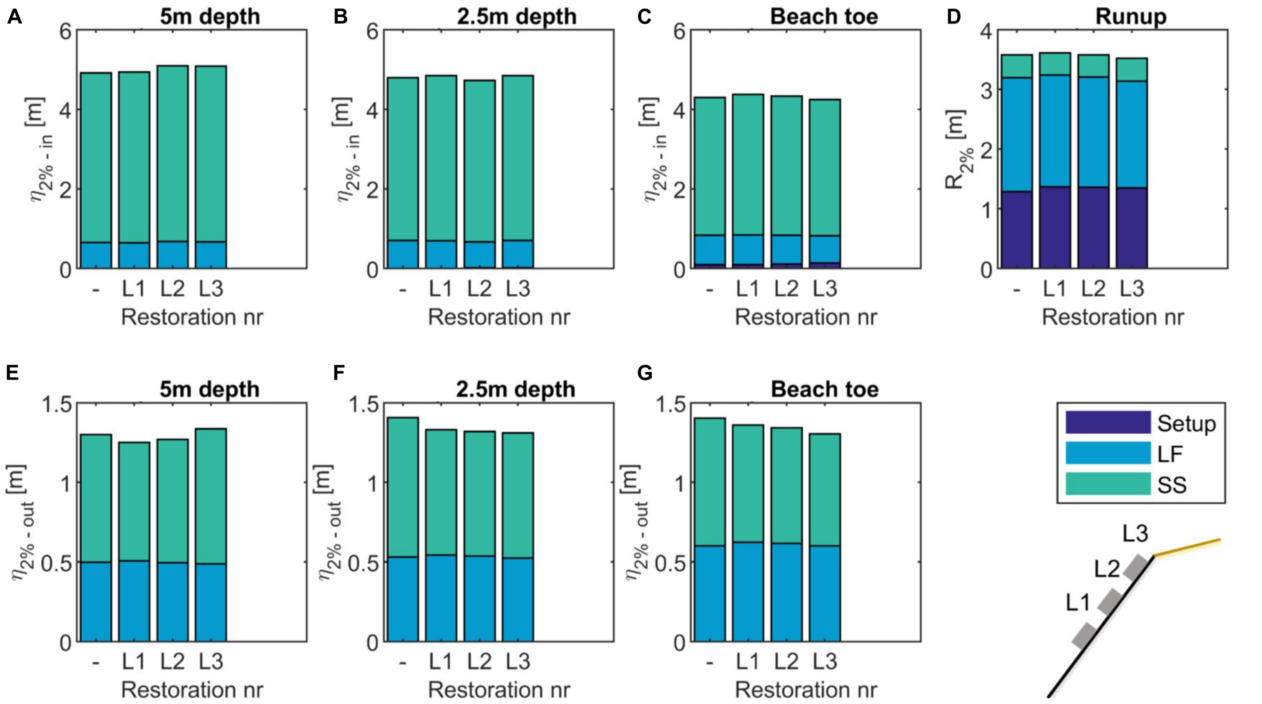

FIGURE 4 | The influence of different restoration configurations on water levels across a linear reef profile. The mean short (SS) wave, long (LF) wave, and setup water level extremes at three locations across the reef: **(A)** Reef crest, **(B)** mid reef flat, and **(C)** beach toe for profiles with no restoration [Restoration nr (-)] and restored profiles (Restoration nr = L1 to L3, corresponding to **Figure 1**) averaged across model variations. **(D)** Mean runup heights. Water levels are separated into incoming **(A–C)** and reflected **(E–G)** water levels.

reflection of both SS and LF waves reduces wave energy reaching the shore. SS wave reflection is clearly identifiable at restoration location F3 of the fringing reef, where just offshore of the reef crest restoration (**Figure 2E**), $\eta_{2\%-SS-out}$ is significantly larger than at the unrestored profile (compare with first bar). The reduction in $\eta_{2\%}$ at the beach toe (**Figures 2–5C,G**), relative to the unrestored profiles, indicates higher dissipation of both SS and LF waves by wave breaking and bottom friction across the restoration due to the decreased water depths above the restoration and enhanced roughness. However, the reduction in SS wave heights also leads to an increase in setup across the restoration. Especially for locations near the current (pre-restoration) breakpoint (i.e., location F3 of the fringing reef, location C2 of the convex reef), radiation stress gradients due to wave dissipation significantly increase, leading to an increase in setup (**Figures 2–5B–D**). Interestingly, although setup and LF wave height variations clearly translate into a change in their runup components (**Figures 2–5C,D**), the SS wave runup is hardly affected by the SS-wave damping. It is hypothesized that the depth-limitation on the SS-wave height causes its runup to be relatively constant, whereas the reduced breakpoint forcing and diminished energy transfer to LF waves by short wave damping induces a significant reduction in the setup and LF wave component of the runup. The SS wave runup is only reduced for cases with a strong reduction in setup (and hence a decrease of the nearshore water depth; **Figure 2–5D**).

Runup Reduction by Coral Reef Restorations

From the $R_{2\%}$ reduction values across different restoration locations for the four reef types, the following two main observations can be made regarding the efficiency of restorations (**Figure 6**). First, runup and thus flooding reduction varies significantly between reef profile types and is dependent on the location and dimension of the restoration on the profile. The coastlines fronted by three-slope profiles are relatively unprotected from wave action in unrestored conditions, as the observed runup is much higher than at fringing and convex reef profiles. This renders these types of reef profiles vulnerable to coastal flooding but also responsive to restoration measures. For the three-slope profiles, mean reductions in runup of 23 to 30% are achieved, which increase as restorations are placed closer to shore. Across linear reef profiles, runup reduction increases significantly for shallower restoration depths, where runup reductions of up to 10% can be achieved, with absolute values significantly higher than across fringing and convex reefs. For restorations across the three-slope and linear reef profiles, restoration efficiency (reduced flooding potential) increases as dimensions (restoration height and width) increase, due to the enhanced dissipation and reflection across the restoration, which leads to a reduction in setup and LF component of the runup. Mean runup reductions were similar between fringing and convex reef profiles and in the range of −1 to 13%, with the

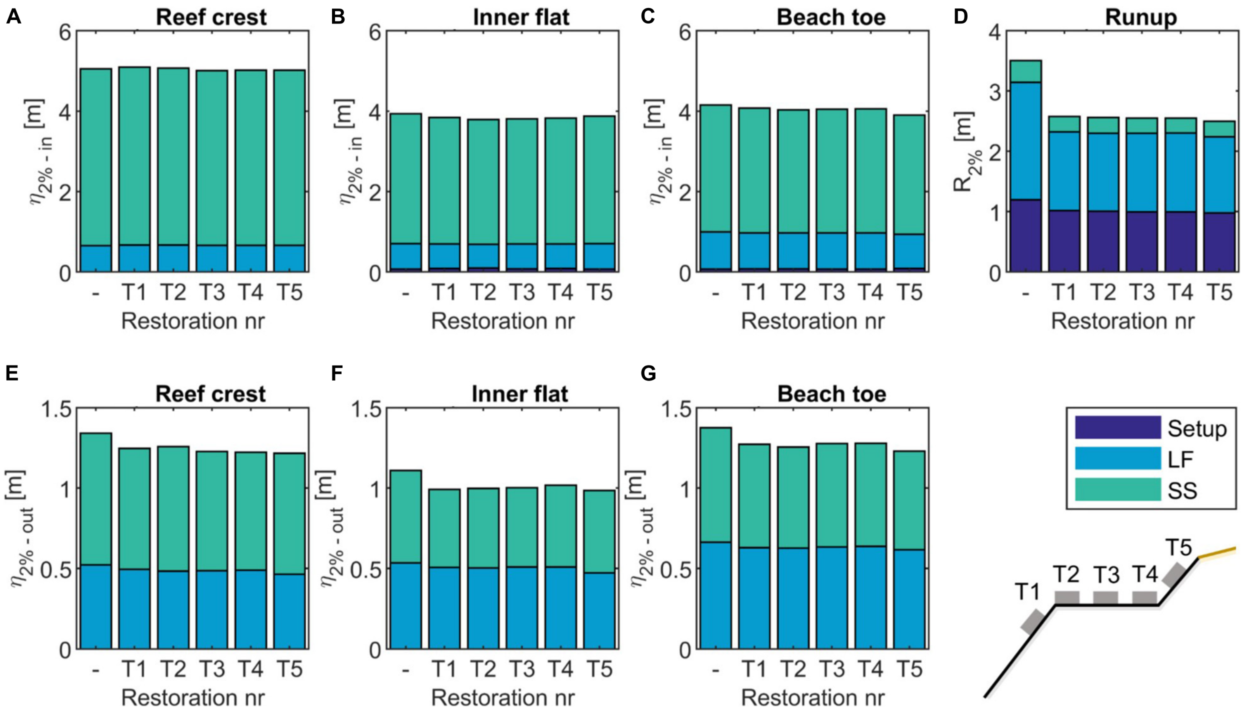

FIGURE 5 | The influence of different restoration configurations on water levels across a three-slope reef profile. The mean short (SS) wave, long (LF) wave, and setup water level extremes at three locations across the reef: **(A)** Reef crest, **(B)** mid reef flat, and **(C)** beach toe for profiles with no restoration [Restoration nr (-)] and restored profiles (Restoration nr = T1 to T5, corresponding to **Figure 1**), averaged across model variations. **(D)** Mean runup heights. Water levels are separated into incoming **(A–C)** and reflected **(E–G)** water levels.

restoration efficiency being strongly controlled by the location and dimension of the restoration. Restorations located near the current (pre-restoration) breakpoint (e.g., locations F3 of the fringing reef profile and location C2 of the convex reef profile) enhance radiation stress gradients and can therefore increase the setup, making them relatively ineffective as coastal protection measures. The amplification of the setup by restoring the reef is aggravated for larger restoration heights. Restorations on the lower fore reef (F1 and F2 of the fringing profile and C1 of the convex reef) can reduce flood risk, although runup reduction is low and can also be negative (relative to the unrestored state), as they are located in relatively deep water with little effect on the wave dynamics. Runup reduction efficiency is highest at the mid-flat, showing mean values of seven and 10% for the fringing and convex profile, respectively. Ideally, the reef flat restoration should be located in between the mid-flat restoration and the inner flat restoration, where wave heights have already been naturally dissipated across the reef, thereby minimizing the additional setup across the restoration. However, the restoration should not be located too close to the inner surf zone where radiation stress gradients are again increased by the restoration and reflection can cause unwanted effects.

Secondly, the runup reduction efficiency highly depends on oceanographic forcing conditions (i.e., the offshore water level, wave height, and wave period). The analysis of runup reduction indicates a strong correlation between the LF and setup component of the runup and these extrinsic parameters, whereas the SS wave runup component is relatively invariable due to the depth-limitation on the SS-wave height. Both the LF wave runup reduction and setup increase across the restoration, as they are proportional to the incident wave height and wave period and are inversely proportional to the depth on the reef. Although enhanced radiation stress gradients (large wave heights, low water depths) across the restoration locally enhance setup, enhanced wave dissipation actually leads to diminished setup near the shore. Across linear and three-slope profiles, this effect translates into a runup reduction that increases for shallower water depths, larger wave heights, and longer wave periods, as a result of nearshore setup and LF runup reduction. For the fringing and convex reef, trends in LF runup reduction are clear, but the total runup reduction is difficult to predict due to the large influence of the setup across these profiles, which is highly sensitive to varying hydrodynamic conditions.

DISCUSSION

Coral restorations are often performed with the purpose of enhancing reef ecology, often with marginal consideration for, and research investments in, their coastal protection value. We demonstrate the positive impact of coral restorations on adjacent coastal flood reduction potential, showing a promising example of a nature-based solution for vulnerable coral reef-lined coasts. Characteristic reef profiles, deduced *via* a statistical analysis, were

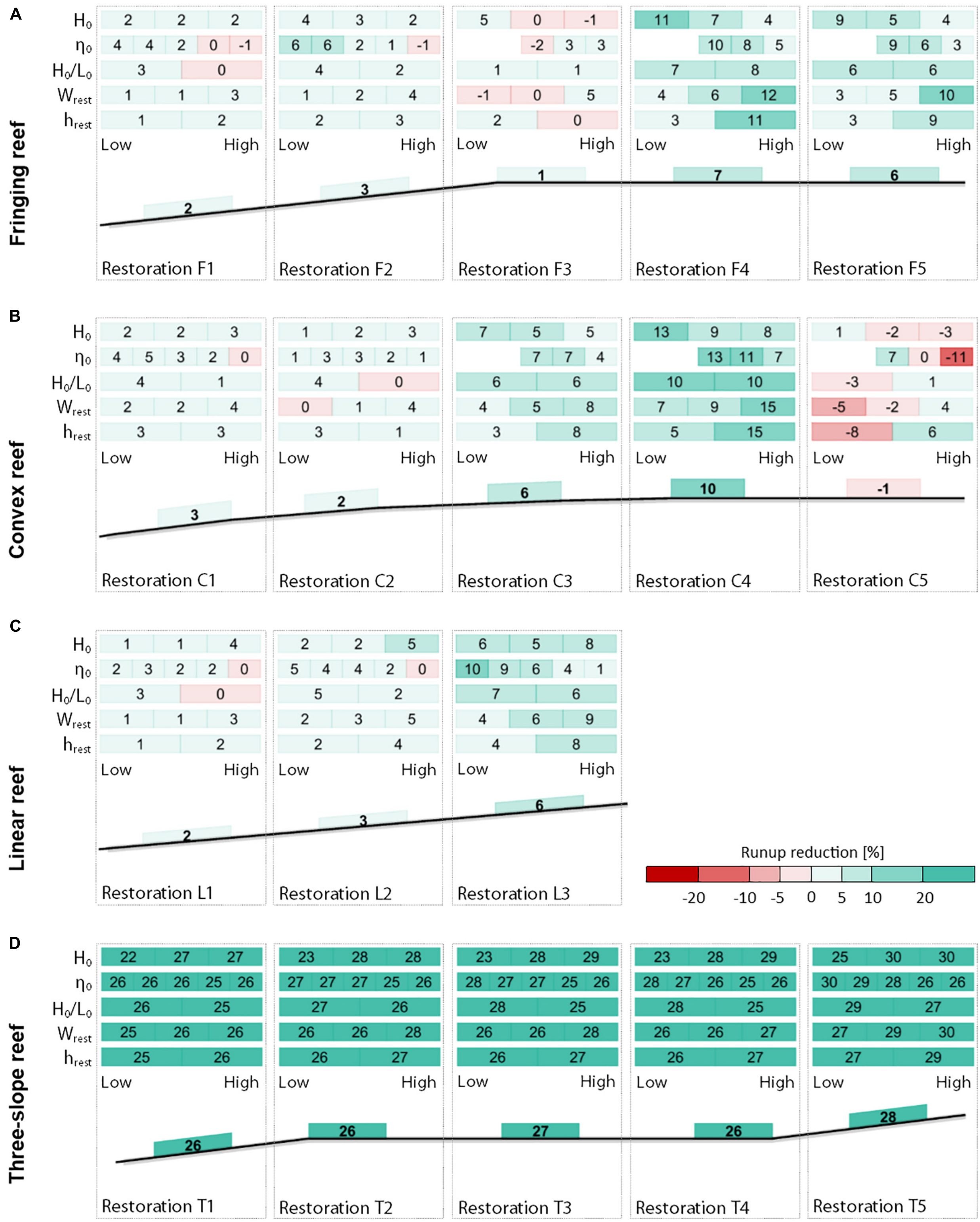

FIGURE 6 | Runup reduction potential for restorations on the four characteristic reef profile types. Mean runup ($R_{2\%}$) reduction values [%] across different restoration locations (1–5) for the **(A)** fringing reef, **(B)** convex reef, **(C)** linear reef, and **(D)** three-slope reef. For each reef type, the mean runup reduction is calculated for filtered input parameters (varying offshore wave heights H_0, water levels η_0, wave steepness values H_0/L_0, restoration widths W_{rest} and restoration heights h_{rest}), in which runup is averaged across all other model variations. "Low" and "High" refer to the minimum and maximum input values for the parameter. The total mean runup reduction is depicted for each restoration location as the value denoted in the illustrated reef restoration blocks.

classified as the fringing reef, the convex reef, the linear reef and the three-slope reef, and were subjected to restorations at different across-shore locations along the profile. Model results indicate that the wave reflection and dissipation across coral restorations can decrease potential coastal flooding up to 30%, the exact reduction efficiency being highly dependent on (1) the reef profile shape, (2) the location of the restoration on the profile, (3) dimensions of the restoration, and (4) hydrodynamic forcing conditions. Fringing and convex reefs feature a reef flat that already acts as a natural wave attenuator, limiting the additional flood risk reduction effect that could be provided by coral restoration. Still, runup reductions of up to 10% can be achieved for shallow restorations located on the reef flat at some distance from the beach toe. For both reef types, the reduction of wave-driven flooding is greater for shallower restorations on the upper fore reef or middle reef flat than for deeper locations on the fore reef. The reef crest restoration of the fringing reef (F3), as well as the inner reef restoration of the convex reef (C1) can actually increase wave-driven flooding as a result of increased mean water levels (C3 or F3) and shoreward reflection of waves at the restoration (C1 or F1). The linear and three-slope profiles leave the coastline relatively unprotected from wave action, rendering them naturally more vulnerable to coastal flooding, but also more receptive to coral reef restoration measures. Average runup reductions of 26–30% are observed for the three-slope profile, and up to 10% for the linear profile, with greatest reduction for shallow restorations near the shore.

This study presents a first attempt at describing and quantifying the beneficial added value of coral reef restorations for coastal hazard risk reduction. However, this approach is not without limitations. First, coral reefs were modeled as highly schematized profiles, where impermeable locally raised bed levels with enhanced roughness imitate reef restorations. In nature, coral transplantations and artificial reef restorations are not impermeable, allowing canopy flow that likely reduces the wave-setup over and reflection at the restoration compared to the impermeable bed case modeled here. Whether simulated hydrodynamic patterns are consistent with observations in nature should be verified with field or laboratory experiments, which are currently not available. Second, one-dimensional reef models neglect two-dimensional effects such as horizontal circulation cells and longshore currents that may balance wave-induced set-up with offshore flow out of channels in the barrier reef (e.g., Lowe et al., 2010). In addition, structure-induced circulation patterns as commonly observed for submerged structures (Villani et al., 2012) are neglected in the 1D approach. Third, the assumption of normal wave incidence is likely to cause an overestimation of the LF-wave height and LF runup component due to the enhanced interactions between individual waves in a 1D approach (e.g., Herbers et al., 1994). However, this limitation is minor as focus is foremost on the relative comparison of flood risk rather than an absolute representation of flood risk. Fourth, estimates of runup on reef-fronted coastlines have scarcely been validated due to the global lack of runup recordings (e.g., Winter et al., 2020). Despite these limitations, model simulations of previously validated XB models (Quataert et al., 2015, 2020) suggest models presented in the present paper give a reasonable estimate of the relative impact of coral reef restoration measures on coastal flood risk.

It is apparent from the results presented here that most of the optimal locations on reefs for restorations to reduce coastal flooding are in shallow, energetic areas such as the upper fore reef and middle reef flat that are typically characterized by physically-robust coral species (e.g., Montaggioni and Braithwaite, 2009). Hence, to increase the coastal hazard risk reduction potential of coral reef restoration, physically robust species of coral need to be grown in nurseries and outplanted to shallow, energetic locations. This, however, contrasts the current practice of more fragile, faster-growing species primarily being grown in nurseries (Levy et al., 2010; Lohr et al., 2015) and outplanted in the field (Bostrom-Einarsson et al., 2020, and references therein). Further, ocean warming and acidification (Pandolfi et al., 2011) as a result of climate change will likely adversely impact coral growth on the reef flat, necessitating outplanting strategies with climate-resilient corals.

An uncertain future under climate change, with a possible increase in storm intensity or frequency, and rising sea levels, mandates the need for improved coastal resilience to hazards through effective adaptation measures. The results of this study suggest that, under storm conditions, the flood reduction potential provided by coral restorations across linear and three-slope profile reefs would likely increase due to the enhanced dissipation across restored reefs. In contrast, the flood reduction potential may decrease across fringing and convex reefs under similar conditions, reducing the overall efficiency of the restoration, although a small reduction in runup may still be achieved. Additionally, rising sea levels could make coral development more viable on the reef flat (Scopélitis et al., 2011), while deeper restorations on the fore reef may become less effective. At present, relatively few corals are typically found on shallow reef flats (relative to fore reef slopes) due to wave energy, thermal tolerances, and occasional exposure to open air during spring low tides. Rising sea levels may actually reduce the impact of these factors on coral growth patterns, which may be especially positive for fringing and convex reef restorations, as they would improve energy dissipation by increasing the hydrodynamic roughness of the reef flat.

Hazard mitigation strategies can help finance reef restoration in different ways. First, post-disaster recovery funding could support restoring coral reefs for coastal defense infrastructure (Beck and Lange, 2016). Second, because coral reefs protect coastal communities, reef restoration could be funded through such mechanisms as pre-disaster hazard mitigation funds (Beck and Lange, 2016). Third, the insurance industry can support incentives for habitat restoration by insuring their coastal protection service (Reguero et al., 2019) or through new resilience insurance mechanisms for coral reef restoration projects (Reguero et al., 2020). By allocating ever-limited funds to well-designed coral restorations, vulnerable coastal areas may receive the much-needed support to restore their adjacent reefs to support tourism, fisheries, and recreation *via* such mechanisms as pre-disaster mitigation, post-disaster restoration funding, and/or insurance. Through the improved understanding of their optimal runup reduction efficiency, coral reef restorations may become a physically and economically viable option for mitigating hazards under a changing climate with benefits for both coral ecosystems and human coastal populations.

DATA AVAILABILITY STATEMENT

The XB-NH+ model input files are available as a NetCDF (*.nc) file at the following location: https://doi.org/10.5066/P991RSFO.

AUTHOR CONTRIBUTIONS

FR, CS, and AD designed research and methodological approach. FR performed the modeling. All authors analyzed the results, performed the visualizations, and co-wrote the manuscript.

FUNDING

This work was supported by the U.S. Department of Interior, U.S. Geological Survey through the Coastal and Marine Hazards and Resources Program's Coral Reef Project under Grant/Cooperative Agreement No. G20AC00009, and the Deltares Quantifying Flood Hazards and Impacts (11205283) and Natural Hazards (11206875) Strategic Research Programs. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

ACKNOWLEDGMENTS

We would like to thank Ben Norris (USGS) for his important insight and useful comments during the preparation of this article. We would also like to thank the two reviewers for their constructive comments, which improved this manuscript.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2021.653945/full#supplementary-material

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Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Environmental Science

ORIGINAL RESEARCH
published: 22 July 2021
doi: 10.3389/fenvs.2021.686077

Estimating Benefits of Nature-based Solutions: Diverging Values From Choice Experiments With Time or Money Payments

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Nature-based solutions (NBS) provide a promising means to a climate resilient future. To guide investments in NBS, stated preference studies have become a common tool to evaluate the benefits of NBS in developing countries. Due to subsistence lifestyles and generally lower incomes, SP studies in developing countries increasingly use time payments as an alternative to the traditionally implemented money payments. It remains unclear, however, how time values should be converted into money values, how the payment affects willingness to pay (WTP) estimates, and how this influence varies across settings with different levels of market integration. We compare the results of choice experiments that use either time or money payments and that are implemented in urban and rural Ghana. The choice experiments target to value different NBS aimed at erosion prevention and other ecosystem service benefits along the highly erosion prone Ghanaian coastline. Time payments are converted into monetary units using two generic wage-based conversion rates and one novel individual-specific non-wage-based conversion rate. We find higher WTP estimates for the time payments. Moreover, we find that the underlying implicit assumptions related to the currently commonly applied generic wage-based conversion rates do not hold. Finally, we find higher levels of market integration and smaller WTP disparities in the urban site, providing evidence that market integration allows for convergence of WTP estimates. These results provide guidance on the accurate estimation of NBS benefits through the implementation of stated preference studies with time payments.

Keywords: discrete choice experiment, economic valuation, nature-based solutions (nbs), time payment vehicle, market integration, ecosystem services, non-market valuation, stated preferences

OPEN ACCESS

Edited by:
Boris van Zanten, World Bank Group, United States

Reviewed by:
Jens Rommel, Swedish University of Agricultural Sciences, Sweden
Daniel Balsalobre-Lorente, University of Castilla-La Mancha, Spain

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Specialty section:
This article was submitted to Environmental Economics and Management, a section of the journal Frontiers in Environmental Science

Received: *29 March 2021*
Accepted: *05 July 2021*
Published: *22 July 2021*

Citation:
Hagedoorn LC, Koetse MJ and van Beukering PJ H (2021) Estimating Benefits of Nature-based Solutions: Diverging Values From Choice Experiments With Time or Money Payments. Front. Environ. Sci. 9:686077. doi: 10.3389/fenvs.2021.686077

INTRODUCTION

Over the years, nature-based solutions (NBS) have increasingly gained interest from both scholars and decision-makers due to the role that these measures can play in the transition to a sustainable and climate resilient future. NBS are defined as measures that aim to protect, sustainably manage and restore ecosystems, thereby addressing societal challenges while providing both human well-being and biodiversity benefits (e.g. Cohen-Shacham et al., 2016; WWF International, 2020). However, current investment in NBS is limited (WWPAP/UN-Water, 2018; Deutz et al., 2020), even though both the physical and cost-effectiveness of NBS have been proven (Ferrario et al., 2014; Narayan et al., 2016; Reguero et al., 2018). Because developing countries are generally more vulnerable to the impacts of climate change and natural hazards (IPCC, 2014; Jongman et al., 2015; Hossen et al., 2019), developing countries may benefit more from NBS. Still, compared to developed countries, progress on transitioning to a sustainable and climate resilient future in developing countries is lagging behind (Hinkel et al., 2012; IPCC, 2014).

To guide investments in NBS, stated preference (SP) studies have become a common tool to evaluate the benefits of NBS (Brouwer, 2008; Bockarjova and Botzen, 2017). SP studies are especially suitable for this purpose since they allow for the valuation of non-marketed goods, such as erosion prevention. The damage cost avoided approach is another method that is commonly applied to estimate benefits related to ecosystem services as erosion prevention. However, non-marketed goods are commonly neglected in applying this approach and one needs ample existing data to be able to apply the approach. The latter is especially challenging in developing countries, where data availability is generally lower. To exemplify, an IMDC (2017) study that adopted the damage cost avoided approach in Ghana identified 11 benefits and describes that reliable data is available for only two of those benefits. Therefore, applying the damage cost avoided method instead of a SP method would increase the risk of underestimating the benefits of NBS.

In SP studies, respondents are asked to make trade-offs between positive (negative) changes in ecosystem services and a payment (compensation). This payment is commonly monetary. Due to subsistence lifestyles and generally lower incomes, a monetary payment complicates trade-offs in developing countries and may therefore lead to issues with the estimation of the willingness to pay (WTP) for NBS. More specifically, using money payments could lead to an underestimation of WTP (Alam, 2006; O'Garra, 2009; Hagedoorn et al., 2020; Meginnis et al., 2020), a failure in accurately representing the preferences of certain groups in society (Alam, 2006), and various methodological problems (Gibson et al., 2016).

Time payments serve as the most popular alternative to money payments (e.g. Gibson et al., 2016; Tilahun et al., 2017; Pondorfer and Rehdanz, 2018; Owour et al., 2019; Rai et al., 2019; Alfredo and O'Garra, 2020; Endalew et al., 2020; Hagedoorn et al., 2020; Meginnis et al., 2020; Navrud and Vondolia, 2020; Van Oijstaeijen et al., 2020). The reasoning is that overall WTP is not only a function of a person's monetary ability to contribute (e.g. money) but also of their non-monetary ability to contribute (e.g. time). Especially in developing countries non-monetary goods can also serve as a means of payment. After all, similar to money, time is subject to a budget constraint and can have high opportunity costs. Moreover, time payments are found to be highly accepted by respondents (e.g. O'Garra, 2009; Abramson et al., 2011; Casiwan-Launio et al., 2011; Rai and Scarborough, 2013; Alfredo and O'Garra, 2020; Girma et al., 2020; Hagedoorn et al., 2020; Meginnis et al., 2020).

Despite these advantages of using time payments, challenges remain especially on the conversion of time into monetary values, thereby facilitating incorporation of time payments in economic analyses. So far, studies mostly apply a generic wage-based conversion approach to convert time values to monetary ones, for instance studies use an average wage value to convert time or apply a fraction to this wage value to estimate a leisure rate as based on Cesario (1976) (e.g. O'Garra, 2009; Casiwan-Launio et al., 2011; Vondolia et al., 2014; Gibson et al., 2016). These studies thereby implicitly assume that the value of time is the same across all respondents, that all respondents would sacrifice the same activity (i.e. wage or leisure time), and that the value of leisure time is equal to a generic fraction of the wage rate[1]. However, all these assumptions can be questioned. Furthermore, ambiguous results from previous studies on WTP disparities resulting from time and money payments raise questions on the drivers and impacts of these disparities (e.g. Casiwan-Launio et al., 2011; Vondolia et al., 2014; Gibson et al., 2016). Expanding the knowledge on the drivers of the WTP disparities from time and money payments can be useful in guiding future payment vehicle use in developing countries.

Research Questions

In order to address the challenges related to time payments in SP studies, and thereby guide future decisions on payment vehicle use in valuation studies, we compare WTP estimates of time and money payments in a developing country context (Ghana). Additionally, we also compare these results across a rural and an urban setting with the aim to examine the effects of market integration on WTP disparities. We implement an identical choice experiment in both a rural and an urban study site, aimed at valuing erosion prevention and related co-benefits. We measure market integration by examining a range of aspects based on different strands of literature. Moreover, we add to the literature and discussion on converting time to money by comparing two traditional generic wage-based conversion rates to an individual-specific non-wage-based conversion rate. The four research questions that we address in this study are as follows:

1) Research question 1: How do WTP estimates from an experiment with a time payment vehicle differ from WTP estimates from an experiment with a monetary payment vehicle?
2) Research question 2: Do the underlying implicit assumptions related to a generic wage-based conversion rate hold when comparing these values to those resulting from a non-wage based conversion rate?
3) Research question 3: How do market integration levels differ across urban and rural settings?
4) Research question 4: How do differences in WTP estimates from time and money payment vehicle experiments correspond to differences in market integration levels?

The results of this study contribute to the existing literature on stated preference studies and valuation of NBS, with a specific focus on a developing country context. Our approach and findings provide useful guidance on how to convert time to money and we provide a discussion on when to use a time or a money payment vehicle. By improving the applied methodology and thus generating more accurate and reliable values for NBS

[1]We refer to *Time-To-Money Conversion* for more detail on the underlying implicit assumptions related to current practices on the time-to-money conversion.

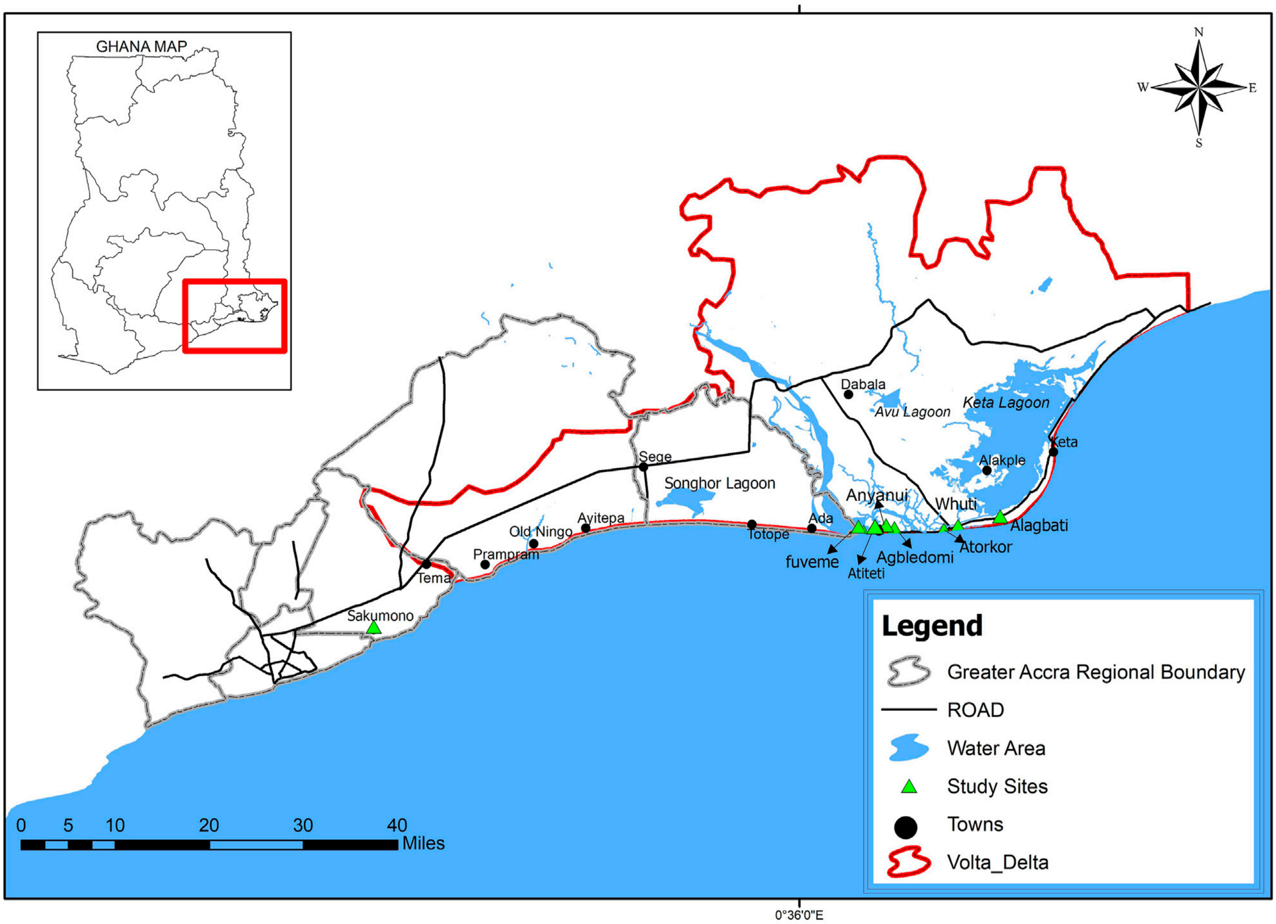

FIGURE 1 | Map with the locations of the study sites.

benefits, the results of our study can be used by future SP studies to improve the quality of economic information provided to the decision-making domain to guide investments in NBS in developing countries.

The remainder of this paper is structured as follows. First, we continue this section by introducing the case study sites. *Literature Review* provides a review of the literature on the conversion of time to monetary values and discusses market integration as potential driver of WTP disparities from time and money payment vehicles. *Methods* describes the data collection and presents the methods. In *Data and Analysis* we describe the data characteristics followed by our approach to analyze the data. *Results* presents the results. Finally, *Discussion and Conclusion* provides a discussion of the results and presents the main conclusions.

Case Study Sites

To compare WTP estimates from time and money payments in a developing country context we selected two coastal study sites in Ghana (see **Figure 1**). The first is the coastal stretch between the communities of Fuveme and Anloga in the Volta delta, a rural site that includes eight smaller communities. The second is the community of Sakumono situated in Tema City, an urban site that includes one larger community. Ghana is a developing country whose coastal areas are changing rapidly. A major issue along the Ghanaian coast is the high erosion rates, to which local geology, human activities and climate change all contribute (Laïbi et al., 2014; Gomez et al., 2020). The ongoing erosion negatively affects coastal ecosystems and puts pressure on local livelihoods that depend on these ecosystems and the services they provide. In order to mitigate these negative effects, several NBS focused on erosion prevention were selected prior to this research, based on meetings with the community leaders, site visits and insights from local researchers and NGO staff. The potential restoration activities include beach nourishment, restoration of coastal lagoons, and recovering mangrove forest in the Volta delta. These restoration activities are expected to prevent erosion and provide other ecosystem service benefits.

In the Volta delta, the selected study site is situated between the Keta lagoon, Volta river and Gulf of Guinee and is thus surrounded by water and mangroves. The people depend on the different ecosystems for fisheries, fuelwood and protection from erosion and floods (The Development Institute, 2016). Due to overharvesting of fish stocks and mangrove wood in addition to ongoing erosion the ecosystems are degrading. Erosion in the Volta delta is occurring at rates between two and 10 m per year (Ly, 1980; Boateng, 2012; Deltares Aqua Monitor, 2019) forcing communities to move away from the coastline (Roest, 2018). Restoring the ecosystems can lead to improvements in the delivery of ecosystem services such as erosion control, coastal protection, fisheries, fuelwood, spiritual values, recreation and tourism opportunities.

In Tema City, a port was built in 1962 to provide cargo services to the surrounding region. Since then the city has grown substantially and local ecosystems have become degraded along the way. Main environmental issues are the coastal erosion rates of one to 5 m per year (Ly, 1980; Boateng, 2012; Deltares Aqua Monitor, 2019) and declining fish stocks (Atta-Mills et al., 2004; ISD, 2018). Adjacent to Sakumono community, the city's largest lagoon is furthermore increasingly polluted and overgrown with grasses that affect the water flow and fish habitat. This directly affects the livelihoods of the community who still largely depend on fisheries. Restoring the ecosystems can lead to improvements in the delivery of ecosystem services such as erosion control, fisheries, spiritual values, recreation, tourism possibilities and water and air quality regulation.

LITERATURE REVIEW

Time-To-Money Conversion

Converting time values into monetary values is required to compare WTP estimates from time and money payments as well as to use the outcomes of a SP study with time payments in cost-benefit analyses. So far, studies mostly apply a generic market wage as conversion rate (Alam, 2006; O'Garra, 2009; Vondolia et al., 2014; Gibson et al., 2016; Meginnis et al., 2020) or estimate a leisure rate based on the study by Cesario (1976) (O'Garra, 2009; Casiwan-Launio et al., 2011). Both approaches can be criticized since they assume that the value of time is the same for all respondents while in reality we often find heterogeneity in wage and leisure rates. Tilahun et al. (2015) therefore apply individual-specific wage information, while Hagedoorn et al. (2020) compose a conversion rate that is based on individual-specific wage information as well as on information about how a respondent spends his or her time on an average day.

Despite these improvements there are still several other problematic issues with wage-based conversion rates. First, in developing contexts it is common that not all respondents earn a wage. These respondents appear in the dataset as a missing value or as if their value of time is equal to zero, an assumption that poorly reflects reality (Lloyd-Smith et al., 2019). Second, for the conversion rate proposed by Hagedoorn et al. (2020) you need a lot of information from respondents, including income information that respondents might not want to provide. Third, composing the wage and leisure-based conversion rates requires making assumptions with potentially large effects on the value of leisure time (Lloyd-Smith et al., 2019; Hagedoorn et al., 2020). Finally, there is a lack of insights in the type of activity that respondents are willing to sacrifice in order to contribute time, and thus whether we should use wage or leisure time as a proxy of opportunity costs.

Alternatively, there are studies that value time using information that is not based on wage data, for instance through modeling approaches (e.g. Jara-Diaz et al., 2008), revealed preferences (e.g. Fezzi et al., 2014), combined revealed and stated preferences (Feather and Shaw, 1999) and stated preferences (Alvarez-Farizo et al., 2001; Larson et al., 2004; Eom and Larson, 2006; Palmquist et al., 2010; Rai and Scarborough, 2013; Czajkowski et al., 2019; Lloyd-Smith et al., 2019; Meginnis et al., 2020). This paper focusses on the stated preference part of this literature. Lloyd-Smith et al. (2019) asked each respondent for their willingness to accept (WTA) a payment for time spent on specific activities via a questionnaire, while others used the results of choice formats that include both time and money attributes to calculate the opportunity cost of time (Alvarez-Farizo et al., 2001; Larson et al., 2004; Eom and Larson, 2006; Rai and Scarborough, 2013; Czajkowski et al., 2019; Meginnis et al., 2020). The observed individual-specific values of time only weakly correlate with wage values and show high degrees of heterogeneity (Czajkowski et al., 2019; Lloyd-Smith et al., 2019).

A review of the literature indicates that most studies that use time payments in developing countries implement separate stated preference questions for both time and money payment vehicles (e.g. Gibson et al., 2016; Khanal et al., 2019; Endalew et al., 2020; Girma et al., 2020; Hagedoorn et al., 2020; Navrud and Vondolia, 2020; Alemu et al., 2021). This research design feature is likely due to the fact that combining both time and money attributes in one choice format rarely suits applications other than transportation and recreation demand studies. Therefore, including a separate stated preference question on the value of time in the accompanying questionnaire provides a way forward in standardizing the conversion of time to money in valuation studies, and reduces the risk of making false assumptions in the investigated research context. These potentially false assumptions include that the value of time is the same across all respondents, that all respondents would sacrifice the same activity, that respondents either earn a wage or that their value of time is 0, and that the value of leisure time can be measured by taking a generic fraction of the wage rate. Until now, such a valuation of time approach is unexplored in the context of payment vehicle use in developing countries.

WTP Disparities From Time and Money Payment Vehicles

Previous studies that convert time values into monetary ones and consequently compare these results to WTP estimates based on money payments provide ambiguous results regarding the difference in WTP estimates. Five studies find higher WTP estimates for the time payment vehicle (Alam, 2006; O'Garra, 2009; Casiwan-Launio et al., 2011; Hagedoorn et al., 2020; Meginnis et al., 2020), four studies find similar time- and money-based WTP estimates (O'Garra, 2009; Vondolia et al., 2014; Tilahun et al., 2015; Gibson et al., 2016) and two studies find lower WTP estimates for the time payment vehicle (Vondolia et al., 2014; Navrud and Vondolia, 2020). When observing this variation in findings two things stand out. First, in the case of O'Garra (2009) and Vondolia et al. (2014) the findings are highly sensitive to the applied conversion rate. Second, since this explanation cannot explain all variation in findings it seems that the specifics of the study sites also affect the WTP disparities. In short, the literature on differences in WTP across time and money payments is sparse and fails to explain what drives the disparities.

Market integration levels are an example of site-specific factors that may affect WTP disparities. Gibson et al. (2016) observe from the literature that the disparities between WTP from time and money payments are greatest in rural sites and argue that relatively lower levels of market integration in these sites may be driving the larger disparities. The reasoning is that the absence of well-functioning labor markets and high transaction costs in some areas may mean that people are not able to sell their labor and therefore allocate more time to leisure or self-employment (Casiwan-Launio et al., 2011; Tilahun et al., 2015; Gibson et al., 2016). This in turn may decrease opportunity costs of time and lead to higher relative preferences for money. Limited access to credit markets may further increase the preference for money over time. Consequently, the willingness to contribute time compared to money will be higher and lead to the divergence of WTP estimates obtained from time and money experiments.

The empirical evidence on the effects of market integration on the divergence of WTP from time and money experiments is limited to three studies (O'Garra, 2009; Gibson et al., 2016; Hagedoorn et al., 2020). Gibson et al. (2016) investigate the differences between WTP from time and money payment experiments in a rural site in Cambodia. In their study they do not find differences in WTP from the two experiments. They argue that the close connection between their rural study site and a nearby urban area precluded any divergence and that future studies should investigate areas more remote from urban centers. In a study in Fiji, O'Garra (2009) finds that people that are employed in an urban setting are willing to pay more in money terms and less in time. Likewise, in a study in Vietnam, Hagedoorn et al. (2020) find that households with more income from wage labor are less willing to contribute time. The results from the latter two studies confirm the reasoning in the literature but more extensive analyses are necessary to confirm the effect of market integration on WTP disparities. Both Gibson et al. (2016) and O'Garra (2009) furthermore transform time values to monetary values using a generic market wage conversion rate and neither of the studies provides a measurement of local market integration levels.

METHODS

We used a discrete choice experiment (DCE) embedded in a household survey which was implemented in the selected study sites as described in the previous section. To develop the DCE and questionnaire we started with an exploratory pre-test survey, followed by a qualitative and quantitative pilot survey, both serving as input and testing of the design of the main study. In the following sections we will discuss the data collection, the design of the DCE, and the household survey.

Data Collection

Data collection took place between October 2018 and April 2019. The pre-test survey was implemented during October 2018 and the pilot survey during February 2019, both among 50 respondents in each study site. The main study was implemented during March and April 2019, for which we interviewed 480 respondents in the Volta delta and 490 in Sakumono community. Based on population data obtained from community leaders we interviewed every 10th household in the rural site and every second in the urban site, thereby ensuring random sampling. Those that participated in the pilot survey were excluded from the main survey. Respondents were randomly assigned to the money or time experiment and evenly divided across both experiments.

Respondents were interviewed at their home and were asked to answer the questions on behalf of themselves as individuals, except for the questions that targeted household food consumption, income and income sources, and resource extraction. We targeted to interview the household head or the partner of the household head. The interviews were executed face-to-face by two teams of each 12 local enumerators. The enumerators were trained for one day before the pre-test survey, two days before the pilot survey and another day before the main study. Before implementation, the DCE and questionnaires were translated into the local language: Ewe in the Volta delta and Twi in Sakumono community. Insyt Esoko software[2] was used to record the interview answers on mobile phones. To maintain a high quality data collection, survey responses were checked upon submission and a continuous feedback loop was established between the enumerators and the principal investigator.

Discrete Choice Experiment

DCE is a stated preference valuation method in environmental sciences that is often applied to value ecosystem services. The main theoretical underpinnings come from the theory of value (Lancaster, 1966) and random utility theory (McFadden, 1974; Hanley et al., 1998). It includes asking respondents to make repeated choices between descriptions of a good or service that are defined by a number of attributes. Johnston et al. (2017) provide guidance for stated preference studies on environmental goods and services. According to their recommendations we started with qualitative and quantitative testing of our survey and DCE among people from the target population. By doing so, we developed a clear baseline and description of the ecosystem restoration activities, avoided behavioral anomalies, and ensured a payment vehicle description that is perceived by respondents as realistic, credible, familiar and coercive. The test procedures served to reach the goal of presenting respondents with an incentive-compatible valuation exercise that involves a plausible consequential decision. Vossler et al. (2012) note that

[2]https://insyt.esoko.com/en/home.

TABLE 1 | Attributes and attribute levels for the discrete choice experiments (one for each payment vehicle)

Attribute	Money payment vehicle			Time payment vehicle		
	# of levels	Levels	BAU	# of levels	Levels	BAU
Erosion control (*in meters erosion*)	4	1; 0.5; 0	2	4	1; 0.5; 0	2
Fish abundance (*in % increase in abundance*)	4	0; 10; 20	−10	4	0; 10; 20	−10
Visitors (*in % increase in visitors*)	4	10; 30; 50	0	4	10; 30; 50	0
Monthly time spent tending and cleaning the restored areas (*in days, 8 h per day*)	--	--	--	5	1; 2; 4; 6	0
Monthly contribution to a community fund from which the ecosystem management is paid for (*in GHC per month*[3])	5	2; 4; 8; 12	0	--	--	--

truthful preference revelation is possible when respondents believe they have at least a weak chance of influencing the decision. To ensure consequentiality of the valuation scenario, the introductory text to the survey and framing used in the DCE laid out the project partners and funding agencies and it was explicitly explained that the answers to the survey and DCE can serve as input for the design of future environmental management plans (see **Supplemental Appendix A** for the included statements). In the design of the questionnaire we follow the recommendations of Johnston et al. (2017) by including auxiliary and debriefing questions as well as numerous questions concerning demographics.

Pre-Test Survey

The pre-test survey served four specific goals. The first goal of the pre-test survey was to investigate the suitability of different payment vehicles. The list of payment vehicles included in the pre-test survey consisted of payment types that are familiar to the communities. The two most credible and realistic payment vehicles proved to be a monthly contribution to a community fund and time spent tending and cleaning the ecosystems. This was measured through respondent scores on trust, acceptability, practicality and coverage for each type of payment (Morrison et al., 2000). The second goal of the pre-test survey was to identify the most important ecosystem services affected by the restoration activities. In both of the selected study sites, the three ecosystem services that are most important are erosion control, fish abundance and visitors (i.e. tourists as well as local visitors). Third, the results of the pre-test also served as input for the levels and framing of the payment vehicles and attributes. Lastly, an initial value of time was estimated by comparing the results of pre-test survey questions that asked for the respondents' maximum WTP via the community fund and the maximum WTP via time contributions (similar to Hagedoorn et al., 2020). This value of time was used to link the levels of both payment vehicles in the DCE design.

Pilot Survey

The main goals of the pilot survey were to test the clarity of the choice questions, credibility and realism of the presented situations, plausibility of the attribute levels, and clarity of the pictograms and the attribute descriptions. We first conducted five informal interviews in each study site. During these interviews we introduced the research, asked basic survey questions and presented a draft version of the experiment. Through these interviews we gained a more qualitative understanding of how the respondents perceive the different aspects of the experiment. Based on this understanding we improved the experiment before the survey was piloted among 50 respondents in each study site, after which further adjustments were made. For instance, during this phase we adjusted the business as usual (BAU) level of the fish abundance attribute from 0% change to a reduction of 10% since this better reflected the respondent's perceptions of the BAU scenario.

Design

The design of the DCEs is presented in **Table 1**. An identical fractional factorial orthogonal design was used for both experiments: one with money payments and one with time payments. The initial generated fractional factorial design included 60 choice cards, after which dominant choices were identified and adjusted. The quality of our pilot survey results was not sufficient to base priors on for the creation of an efficient design, but we did learn that respondents generally perceive all changes in ecosystem services as positive and payments as negative. Based on this, a choice was identified as dominant if one option in a choice card had higher levels for all ecosystem services attributes and a lower level for the payment attribute as compared to the other option on that choice card. There were five of such choices, for which we switched the payment levels across the two options and thereby eliminated the dominant choice. The 60 choice cards were divided over six versions, so that each respondent was asked to answer ten choice questions. Besides a BAU option, each choice card included two management options A and B. These management options describe situations in which the ecosystem restoration activities are implemented and managed by the households in the community. The attributes are described by four levels and the payment vehicle by five. The levels that are included in the BAU option did not appear in any of the management options. The levels of the attributes were selected based on current erosion and fisheries trends and changes in visitor rates that were perceived as plausible by the respondents, judging by the pre-test and pilot survey results. The levels of the payment vehicles were based on the results of the pre-test and pilot survey and are related by the value of time obtained from the pre-test survey results. The payment vehicles are

[3]1 USD = 0.77 GBP = 5.42 Ghana Cedis (GHC) during the time of the study.

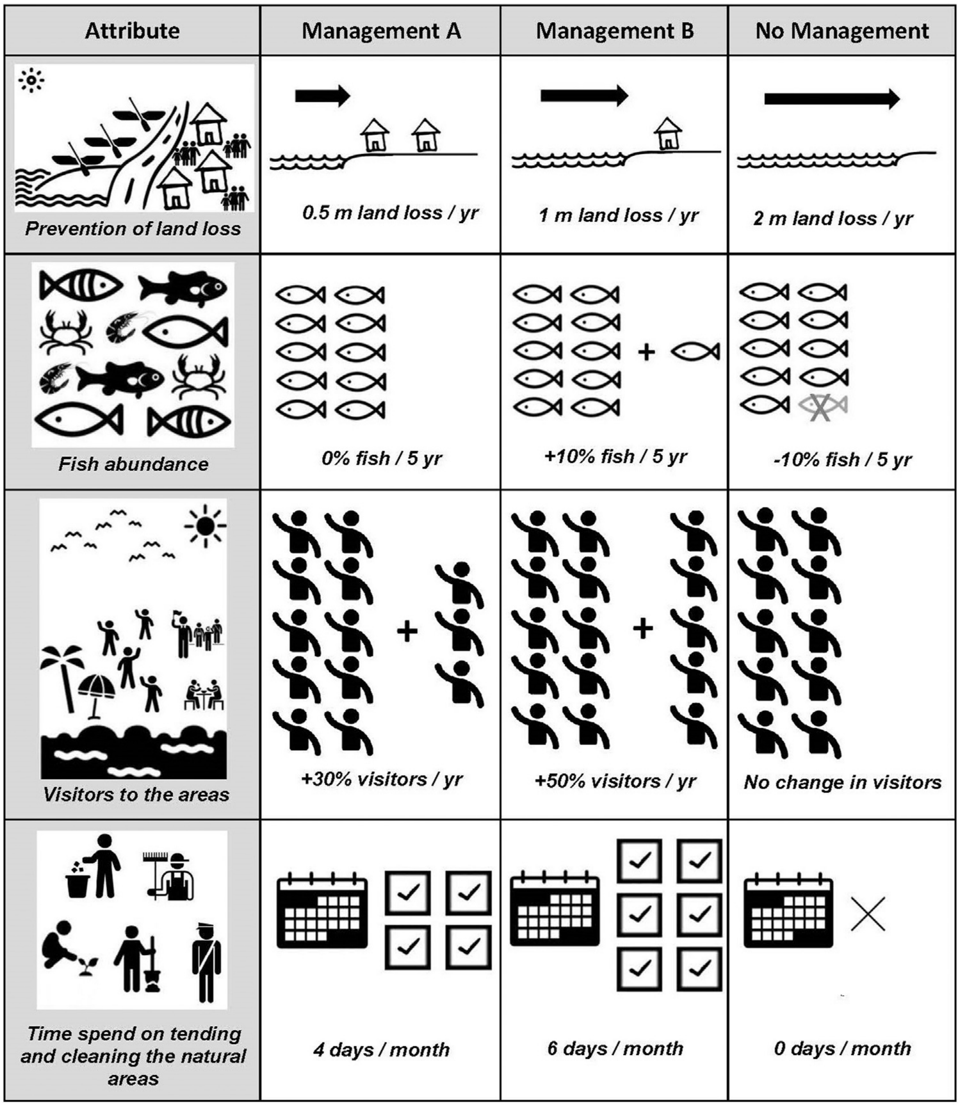

FIGURE 2 | Example choice card for the time payment vehicle, used in both the rural and urban study site.

presented to the respondents as coercive and are similar in terms of framing. This implies that the tasks that would have to be performed would otherwise be paid for with the money collected through the community fund. These tasks include cleaning, guarding, building fences or look-outs, enforcing regulations and planting trees. Example choice cards for both experiments are shown in **Figure 2** and **Figure 3**.

Household Survey

The questionnaire that was used in the household survey consisted of six main sections covering 1) ecosystem use and environmental perceptions, 2) DCE and 3) DCE debriefing, 4) participation in environmental projects, 5) risk perceptions and 6) demographics. The survey was developed in close cooperation between local and international NGOs and universities. Improvements were made based on the results of the pre-test and pilot surveys. The questionnaire is identical for both study sites, except for the questions on mangrove ecosystem use due to the lack of mangroves in the urban site.

Time-To-Money Conversion Rates

In this study we compare three different time-to-money conversion rates. The first two conversion rates are generic wage-based conversion rates. These conversion rates serve as a default that we compare to the third conversion rate. We base the default rates on common practices in the current literature and the study of Vondolia et al. (2014) in particular due to the proximity of our study sites. The third conversion rate is inspired by the approach of Lloyd-Smith et al. (2019) and includes individual-specific values of time that are not based on wage information.

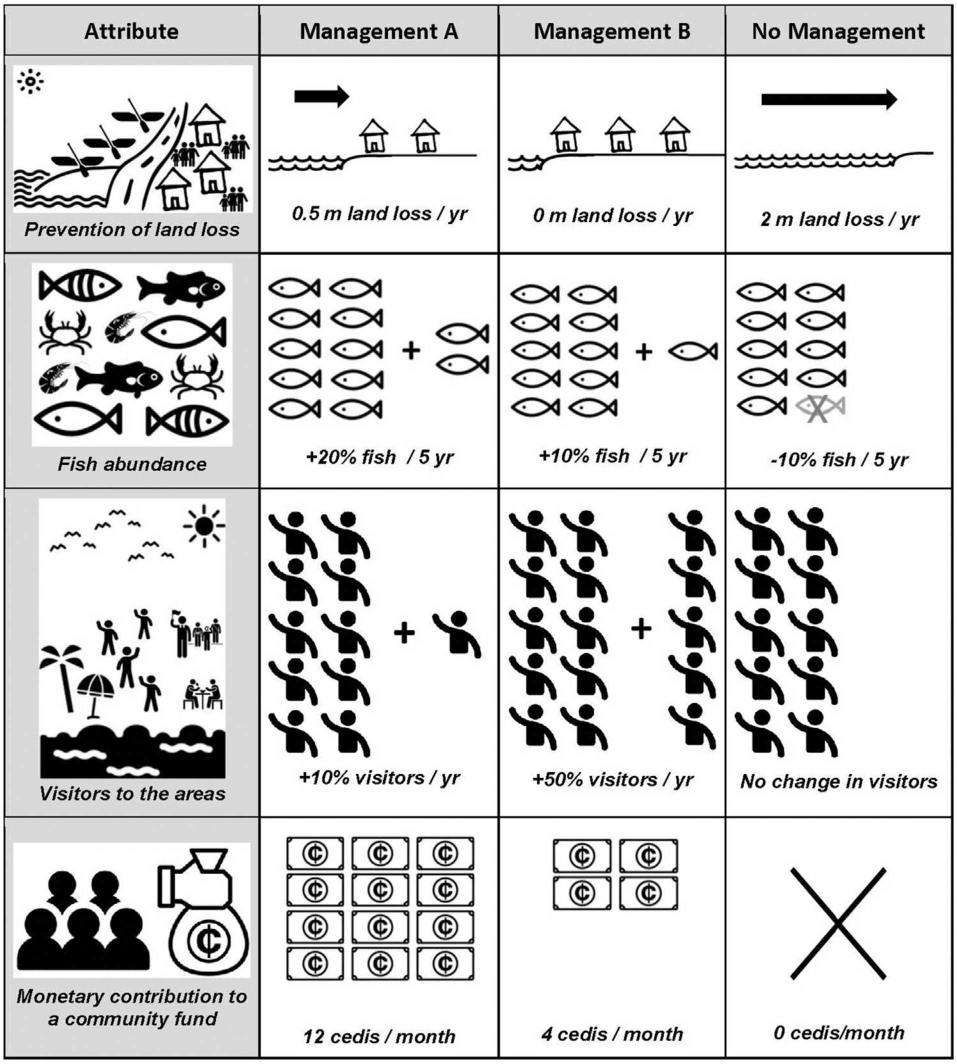

FIGURE 3 | Example choice card for the money payment vehicle, used in both the rural and urban study site.

Conversion Rate 1: Minimum Wage (Time$_{MW}$)

The minimum wage rate in Ghana during 2019 equaled 10.65 GHC for an 8-h work day. For this conversion rate this value is applied to convert the time values, similar to several previous studies (Alam, 2006; O'Garra, 2009; Vondolia et al., 2014; Gibson et al., 2016).

Conversion Rate 2: Sample Earnings (Time$_{SE}$)

For this approach we use data collected through the household survey on wages and hours worked per week to calculate each respondent's wage for an 8-h day spent on wage labor. We also use data collected on trade profits and hours spent on trading per week to calculate each respondent's profit for 8-h of trading. However, more than half of the respondents either does not earn a wage or does not make any money through trading. Therefore, we combine the variables on wages and trade profits to estimate an individual's opportunity cost of time and refer to this new combined variable as *earnings*. If a respondent only earns income from wages and not from trade profits we include the value of an 8-h day spent on wage labor in the *earnings* variable. If a respondent only earns income from trade profits and not from wage labor, we include the value of an 8-h day spent on trading in the *earnings* variable. For those that earn income from both wage labor and trade profits we include the highest value in the *earnings* variable. In the rural site 65% of respondents reported earnings from either wages or trade profits and is thus included in the *earnings* variable, this number equals 70% in the urban site. Since this still means we are missing large portions of our samples we apply a generic value instead of individual-specific values. To estimate the generic value of time, we eliminate those respondents that did not report any earnings (i.e. those that do not earn income from wages or trade profits) and take the median of the remaining sample as the time-to-money conversion rate, similar as Vondolia et al. (2014) did for wages only.

Conversion Rate 3: Non-wage-based Individual Value of time ($Time_{IVoT}$)

To compose the individual-specific non-wage-based value of time we used the approach of Lloyd-Smith et al. (2019) as inspiration and ask respondents for the compensation they would require for an 8-h day of working on the restoration projects. The study by Lloyd-Smith et al. (2019) was conducted in a different context (i.e. fishing trips in the United States) and therefore we had to adjust the approach to fit the Ghanaian context. The main adjustment relates to the use of a stochastic payment card. During the testing phase we learned that working with such a payment card causes difficulties due to low familiarity with the use of probabilities within the Ghanaian communities (see also Navrud and Vondolia, 2020). Therefore, we decided not to use a stochastic payment card but to include an open-ended question format instead. Due to practical reasons (i.e. limited internet and otherwise increased complexity on the enumerators' side) we were not able to include other types of format, such as the double-bounded dichotomous choice format. Adopting an open-ended format over a stochastic payment card means that we simplified the valuation approach and increased the potential for methodological issues. Nonetheless, we believe that this is a small yet valuable step forward with regard to conversion rates in developing countries. Our approach is similar to Lloyd-Smith et al. (2019) in that we describe the formulated activities as if they would comprise a part-time job. We also included a "yes/no filter" question to allow respondents to opt-out, an option that was also embedded in the stochastic payment approach of Lloyd-Smith et al. (2019). This "yes/no filter" question furthermore prevented people from stating unrealistically high values as a result of protest beliefs, an issue that we identified during the test phases when we did not include a "yes/no filter" question which guided people that simply did not want to contribute time to state extremely high values out of protest. As a result, the following question formulation was included in the main study.

Imagine a situation in which an environmental management project is implemented in your community. This could be the replanting of mangroves, deepening and cleaning of the lagoon, defending the beaches so that they do not decrease in size, and keeping all areas, including the sea, waste free. You and other community members could be asked to work between 1 and 6 days per month on this project, after implementation, to ensure that the natural areas will remain in good condition after these measures are taken. This work includes tasks such as picking up trash, guarding the natural area, building fences or look-outs, enforce regulations and planting trees.

Would you be willing to conduct this work?

(Answer: yes or no)

If yes: There might be some compensation available for this work, as if it would be a part-time job. In that case, how much would you want to be paid for one day of such work (8 h)?

(Answer: in GHC per day)

In this question formulation, the descriptions match the framing in the DCE. The question was presented to the respondents in a later stage of the questionnaire, after the DCE and DCE debriefing questions. The results of the second question were used to convert time to money values for each respondent. For those that answered "no" to the first question we did not obtain a value of time. This valuation approach proved to be efficient since it only requires two additional survey questions, compared to a long list of questions such as in Hagedoorn et al. (2020). The approach is also effective in that each respondent is able to express a value of time, also those that do not earn a wage or trade profits (over 30% in our samples) and would therefore otherwise be recorded as a missing value and thus excluded from the analysis. Moreover, the observed value of time is specifically related to the activities that the respondents are asked to conduct in the DCE. We included additional questions related to the certainty of the stated value of time and in regards to which current activity would be given up.

Market Integration Measures

Currently there is no consensus in the literature on whether a single measure can be applied to accurately represent market integration levels, and if yes what that measure should be. Studies conducted in different contexts and with different purposes have used different market integration measures. For example, the literature on Amazon tribes' market integration focuses more on selling resources and labor division whereas studies that are applied in a wider spectrum of locations focus more on participation in, and income from, wage labor and trade profits (e.g. Godoy et al., 2010; Ensminger and Henrich, 2014). Therefore, we included multiple survey questions that measure variables that are related to market integration, based on different strands of literature. First, the studies conducted in the Amazon rainforest aim to investigate indigenous people's market integration by applying measures including the likelihood of requesting a loan, fraction of income from wage labor, and the sale of natural resources (Godoy et al., 1997; Godoy, 2001; Godoy et al., 2005; Godoy et al., 2010; Vasco and Siren, 2016; Vasco et al., 2017). The latter measure is also found in the behavioral economics literature (e.g. Siziba and Bulte, 2012), and in the experimental economics literature (e.g. Ensminger and Henrich, 2014). Ensminger and Henrich (2014) conducted a large-scale study across all continents except for Europe and focused also on market integration measures such as the frequency of engagement in wage labor and trade, the amount of income from wage labor and trade profits, and the percentage of food bought within the household. Finally, the accumulation of durable goods is regarded to be a result of increased market integration (Boughton et al., 2007; Godoy et al., 2010). Based on this literature review, we developed a range of market integration measures as presented in **Table 2**.

DATA AND ANALYSIS

In the following sections, we first present the data characteristics. This chapter then describes the specifications of the models that we apply to the collected data and concludes with the statistical analysis for the differences in urban and rural WTP disparities.

Data Characteristics

Prior to the analysis we excluded protesters from the dataset. They were identified by their choice to select the opt-out in all of

TABLE 2 | Market integration measures

Market integration measure	Measured in …	Based on …
1] Fraction of income from wage labor	% of household income from wage labor	Vasco et al. (2017), Vasco and Sirén (2016), Vasco et al. (2017), Godoy et al. (2010), Godoy et al. (1997)
2] Frequency of engagement in wage labor	Days of wage labor per week	Ensminger and Henrich (2014)
3] Income from wage labor	Monthly household income from wage labor in GHC	Ensminger and Henrich (2014)
4] Fraction of income from trade (not of self-produced goods)	% of household income from trade (not of self-produced goods)	Ensminger and Henrich (2014)
5] Frequency of engagement in trade (not of self-produced goods)	Days of trade per week	Ensminger and Henrich (2014)
6] Income from trade (not of self-produced goods)	Monthly household income from trade in GHC	Ensminger and Henrich (2014)
7] Food that is bought in the household	% of food bought from the store or market	Henrich et al. (2010); Ensminger and Henrich (2014)
8] Sale of natural resources (self-produced goods)	Number of visits to the market (per year) to sell self-produced goods	Siziba and Bulte (2012), Ensminger & Henrich (2014), Vasco et al. (2017), Godoy et al. (2010), Godoy et al. (2005)
9] Likelihood of requesting a loan	Dummy variable for if the household has requested a loan before or not	Vasco et al. (2017)
10] Accumulation of durable goods	Number of durable goods	Boughton et al. (2007), Godoy et al. (2010)

TABLE 3 | Data characteristics for the three samples (money payments, time payments, converted time payments sample) in each area

Variables	**Description**	**Rural area**			**Urban area**		
		Sample values: mean; median (sd)					
		Money payment, *N* = 232	**Time payment, *N* = 239**	**Converted time payment, *N* = 186**	**Money payment, *N* = 245**	**Time payment, *N* = 240**	**Converted time payment, *N* = 193**
Age	Age of respondent	44.97; 40.00 (12.66)	46.36; 40.00 (12.93)	47.53; 50.00 (12.82)	42.79; 40.00 (12.95)	43.19; 40.00 (11.27)	42.77; 40.00 (11.48)
Education level	Education of respondent (1 = no formal education, 2 = primary, 3 = middle, 4 = secondary, 5 = vocational/technical, 6 = post middle/secondary, 7 = university)	3.64; 3.00 (1.86)	3.68; 3.00 (1.81)	3.66; 3.00 (1.78)	3.04; 3.00 (1.57)	3.15; 3.00 (1.61)	3.10; 3.00 (1.48)
Gender	Gender of respondent (male = 1, female = 0)	0.64; 1.00 (0.48)	0.60; 1.00 (0.49)	0.61;1.00 (0.49)	0.55; 1.00 (0.50)	0.54; 1.00 (0.50)	0.55; 1.00 (0.50)
Household size	Number of people in the household	6.17; 6.00 (2.77)	5.90; 5.00 (2.77)	6.11; 5.50 (2.80)	5.52; 5.00 (2.78)	5.69; 5.00 (3.10)	5.91; 5.00 (3.28)
Income	Monthly household income in GHC (continuous based on averages of categories)	920; 900 (545)	930; 900 (572)	928; 900 (581)	1,267; 900 (901)	1,293; 1,250 (858)	1,275; 1,250 (851)
Credibility of the valuation scenario	Level of agreement (Likert scale, 0–10) with the statement: "I believe the changes shown in the experiment can take place in reality"	7.21; 7.00 (1.96)	7.17; 7.00 (2.15)	7.39; 7.00 (2.07)	6.17; 6.00 (2.11)	6.33; 6.00 (2.31)	6.47; 7.00 (2.22)
		Sample values: mean; median (sd) *N* = …					
Daily wage	Daily wage (8 h) of respondent	63.51; 40.00 (83.20) *N = 103*	66.10; 47.33 (83.71) *N = 106*	72.85; 50.00 (92.32) *N = 83*	116.29; 80.00 (103.28) *N = 106*	117.00; 95.98 (90.71) *N = 116*	120.21; 100.00 (92.80) *N = 95*
Daily trade profit	Daily trade profit (8 h) of respondent	67.57; 40.00 (89.89) *N = 94*	87.27; 40.87 (143.85) *N = 113*	97.83; 45.56 (159.04) *N = 90*	177.68; 67.43 (205.75) *N = 121*	187.96; 80.00 (218.43) N = *121*	193.26; 80.00 (219.44) *N = 101*
Daily earnings	Earnings (i.e. from wages or trade profit) from an 8 h work day of respondent	76.21; 48.00 (95.84) *N = 147*	86.73; 50.00 (133.82) *N = 158*	95.48; 50.00 (147.99) *N = 126*	152.61; 66.67 (188.93) *N = 169*	154.38; 60.00 (193.03) *N = 175*	160.17; 55.17 (197.13) *N = 142*
Time IVoT	Individual value of time for 8 h of respondent	49.35; 50.00 (28.81) *N* = 167	50.65; 50.00 (28.61) *N* = 186	50.65; 50.00 (28.61) *N = 186*	51.63; 50.00 (25.00) *N* = 206	56.16; 50.00 (24.43) *N* = 193	56.16; 50.00 (24.43) *N* = 193

the presented choices combined with their answer to a debriefing question that focused on the reason for this choice. If the reason was either a lack of responsibility, lack of trust, or unwillingness to weigh the different attributes against each other we excluded the respondent (e.g. Meyerhoff and Liebe, 2010; Meyerhoff et al., 2014). Nine protesters were identified in the rural sample, of which three in the time sample and six in the money sample. In the urban sample, five protesters were identified in the time sample and zero in the money sample. For the calculation of $\text{Time}_{\text{IVoT}}$ we excluded another ~20% of the respondents from both the rural and urban time samples since they stated that they do not want to work on the restoration projects, answering "no" to the filter question. The datasets without these respondents are referred to as "converted time payment samples".

The key characteristics of the data are presented in **Table 3**. Applying Kruskal-Wallis and Chi-square tests we found no significant differences in any of the variables included in **Table 3** between the samples that either answered to money or time payments in both sites. In the remainder of this paper we compare the money payment samples with the converted time payment samples. In comparing these samples, we only find a higher age in the rural converted time payment sample compared to the rural money payment sample. Furthermore, we measure the perceived credibility of the valuation scenario through a follow-up question. More specifically we asked respondents for their level of agreement with the statement "I believe that the changes shown in the experiment can take place in reality". The results of this follow-up question indicate that on average the respondents perceive the valuation scenario as credible, which furthermore suggest that they perceive their choices as consequential.

Model Specification

We estimated five models for each study site. One for the money payments, one for the time payments, and one for each of the three different conversion rates. The conversion of time values to money values was performed before model estimation by applying the three time-to-money conversion rates to the time values in the converted time payment sample dataset. The converted time attribute was then included in the model instead of the original time attribute. The converted time payments sample dataset is used for all conversion rates to ensure comparability across these model results and to exclude uncertainties related to the answers provided by respondents who stated that they do not want to contribute time but did answer to choice cards that included time contributions. We convert the time values before model estimation to be able to also run the Krinsky and Robb (1986) procedure, and obtain the information necessary for the statistical comparison of differences in WTP across the urban and rural study site as described in the next section.

To allow for preference heterogeneity and obtain coefficient estimates at the individual level, we analyzed each choice dataset using a random parameters logit (RPL) model (Train, 2003). In this model the probability P that individual i chooses alternative j out of $k = 1 \ldots K$ alternatives is equal to:

$$P_{ij} = \int \left[\frac{\exp\left(\beta_i X_{ij}\right)}{\sum_{k=1}^{K} \exp\left(\beta_i X_{ik}\right)} \right] \Delta\left(\beta_i \middle| b\right) d\beta_i, \forall j \in K, \tag{1}$$

in which **X** is a vector with choice attributes and **β** is a vector with attribute parameters to be estimated by the model. These parameters vary per respondent (hence $\boldsymbol{\beta}_i$) with probability density $\Delta(\boldsymbol{\beta}_i \mid \mathbf{b})$. This density can be a function of any set of parameters, but in our case it represents the mean and variance of **β** in our sample. Because the model in **Eq. (1)** has no analytical solution, simulations are needed with which draws are taken from a pre-specified distribution for each $\boldsymbol{\beta}_i$. In our models all attributes are included as continuous variables and for each attribute 3,200 Halton draws are taken from the triangular distribution; number of draws are based on Czajkowski and Budziński. (2019). We use triangular distributions instead of normal distributions to allow for clearer visualization of the WTP distributions and to ensure finite coefficient bounds for all attributes and the payments, which is of relevance for the analysis described in *Statistical Approach to Compare WTP Disparities Across the Study Sites* as well as for obtaining finite moments for the WTP estimates (Rai et al., 2015). However, the main patterns and conclusions are very similar for both distributions.[4] For erosion control the preferences are restricted to negative values since the attribute is coded as increases in erosion for which negative preferences can be expected. For increases in fish abundance positive preferences can be expected and thus here the draws are restricted to positive values. For visitors there is no restriction on preferences since positive as well as negative preferences can be expected related to increases in tourist numbers. The payment vehicle values are redefined to be the negative of the variable and thereafter included in the model with a lognormal distribution and the standard deviation restricted to 0, following the newest model to estimate WTP from Carson and Czajkowski (2019). Carson and Czajkowski (2019) suggest this approach to overcome the problem that exists with the ratio of coefficients approach to calculate WTP, being that it results in an undefined standard error for WTP.

The Krinsky and Robb (1986) procedure was applied to obtain 95% confidence intervals of mean WTP estimates. By doing this for each of the five models we obtained values for $\text{WTP}_{\text{money}}$ and $\text{WTP}_{\text{time(days)}}$ as well as for the converted time payment samples resulting in $\text{WTP}_{\text{time(MW)}}$, $\text{WTP}_{\text{time(SE)}}$ and $\text{WTP}_{\text{time(IVoT)}}$. Furthermore, respondent-specific parameter estimates were used to estimate WTP for each respondent for each attribute for all estimated models. The WTP values on respondent level are used to visualize and statistically compare the WTP distributions by applying Mann-Whitney U tests.

Statistical Approach to Compare WTP Disparities Across the Study Sites

We statistically compare the differences in WTP from time and money payments between the rural and urban sites to identify in which study site WTP disparities are larger. It was not possible to ask respondents to do both time and money choice experiments, since this would increase the cognitive complexity of the survey

[4]Results of the analyses with a norma distribution are availabe on request from the authors.

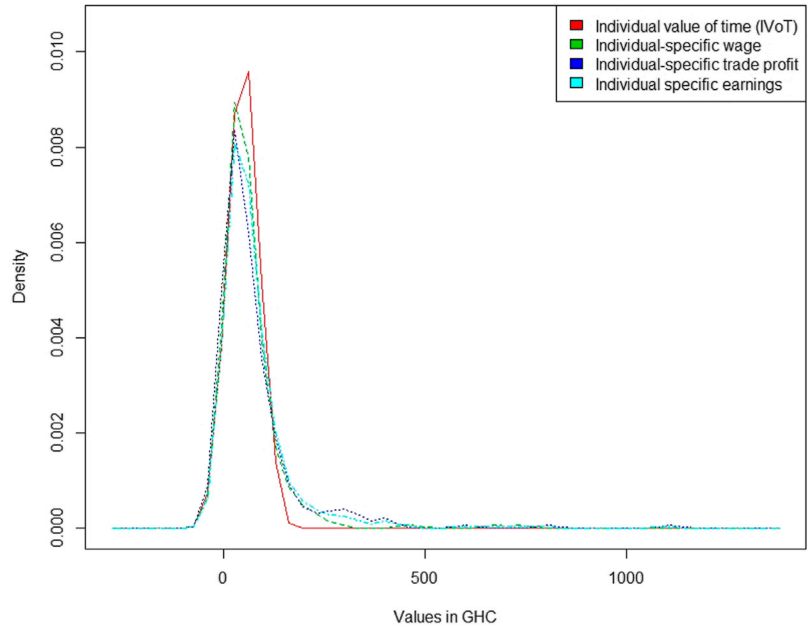

FIGURE 4 | Kernel density plot of $\text{Time}_{\text{IVoT}}$ and individual-specific wage, trade profit and earning values in the rural study site.

too much (based on observations and assessments made during the pre-test and pilot stages). We are therefore not able to compare differences in $\text{WTP}_{\text{money}}$ and time-based WTP on a respondent level. However, through a combination of statistical methods we can still statistically compare WTP disparities across the rural and urban study sites in a meaningful way. We describe the steps taken briefly below, and provide a more extensive description in **Supplemental Appendix B**.

1) Take 10,000 random draws from the triangular distributions of attribute coefficients obtained from the RPL models. By taking into account the full distribution of the attribute coefficients, as an alternative to only the mean estimates, we include as much information as possible in our analysis.
2) Calculate a WTP value for each draw for all attributes.
3) Use the WTP values to calculate percent differences (*PDs*) between WTP estimates derived from the money experiments and from the experiments with different time-to-money conversion rates.
4) Apply Mann-Whitney U tests to statistically compare the *PD*s between the rural and urban site, per attribute. We apply a non-parametric test here since we do not know the distribution and standard error of the *PDs*.

RESULTS

Time-To-Money Conversion

As described earlier, we calculated three different conversion rates. The first conversion rate, Time_{MW}, is based on publicly available information on the Ghanaian minimum wage equaling 10.65 GHC per day. For the second conversion rate, Time_{SE}, data from the household survey resulted in a median sample earnings of 50 GHC per day in the rural study site and 60 GHC per day in the urban study site. The distributions of Time_{SE} are statistically different across the study sites (Mann-Whitney U test, p = 0.001). The estimated values of time for the third conversion rate, $\text{Time}_{\text{IVoT}}$ resulted in a median of 50 GHC in both study sites. The distributions of $\text{Time}_{\text{IVoT}}$ are not statistically different across the study sites (Mann-Whitney U test, p = 0.123). The results of a follow-up question reveal that respondents feel certain about their stated individual value of time (i.e. average of 4.48 out of 5 in the rural site and 4.34 out of 5 in the urban site). To be able to contribute time to the environmental projects, most respondents would give up leisure time (urban: 50%, rural: 68%), followed by subsequently housework (urban: 22%, rural: 9%), and income generating activities such as wage labor (urban: 3%, rural: 9%), trading (urban: 12%, rural: 8%) and fishing or farming (urban: 6%, rural: 6%). In the urban study site, compared to other activities the distribution of the values of time are significantly different for wage labor (mean: 65 GHC) and for leisure (mean: 50 GHC) (Mann-Whitney U test, p = 0.098, p = 0.008). In the rural study site, compared to other activities the distribution of the values of time are significantly different for fishing and farming (mean: 33 GHC) (Mann-Whitney U test, p = 0.008).

Figure 4 and **Figure 5** visualize the relation between $\text{Time}_{\text{IVoT}}$ and the individual-specific values underlying $\text{Time}_{\text{SE,}}$ including individual-specific wages, trade profits and overall earnings. We find positive correlations between $\text{Time}_{\text{IVoT}}$ and the individual-specific wage, trade profit and earnings variables in the rural site (Spearman's Rho = 0.188 with p = 0.019, Spearman's Rho = 0.156 with p = 0.060, Spearman's Rho = 0.181 with p = 0.006, respectively). In the urban site we find a positive correlation between $\text{Time}_{\text{IVoT}}$ and individual-specific trade profits (Spearman's Rho = 0.172 with p =

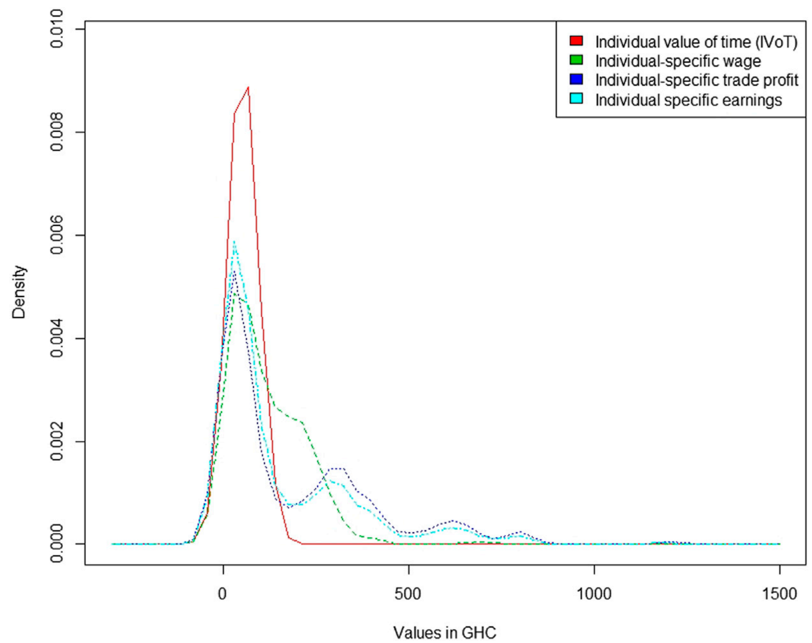

FIGURE 5 | Kernel density plot of $\mathrm{Time}_{\mathrm{IVoT}}$ and individual-specific wage, trade profit and earning values in the urban study site.

TABLE 4 | Results of the RPL models for the time and money experiments in both study sites

Attribute	Rural experiments				Urban experiments			
	Money payment vehicle		Time payment vehicle		Money payment vehicle		Time payment vehicle	
	Coefficient	SE	Coefficient	SE	Coefficient	SE	Coefficient	SE
Means of random parameters								
Erosion control	−1.917***	0.081	−2.179***	0.093	−0.368***	0.055	−0.438***	0.053
Fish abundance	0.021***	0.004	0.030***	0.003	0.048***	0.003	0.053***	0.003
Visitors	0.003*	0.002	0.005**	0.002	0.005***	0.002	0.007***	0.002
Time payments	--	--	−1.737***	0.083	--	--	−2.504***	0.165
Money payments	−2.684***	0.096	--	--	−3.492***	0.196	--	--
ASC opt-out	−30.970***	7.791	−28.744***	6.490	−10.870***	2.171	−6.789***	2.048
Standard deviations of random parameters								
Erosion control	1.917***	0.081	2.179***	0.093	0.368***	0.055	0.438***	0.053
Fish abundance	0.021***	0.004	0.030***	0.003	0.048***	0.003	0.053***	0.003
Visitors	0.010	0.024	0.022**	0.011	0.005***	0.002	0.034***	0.006
Time payments	--	--	0.0	--	--	--	0.0	--
Money payments	0.0	--	--	--	0.0	--	--	--
ASC opt-out	34.861***	8.185	33.315***	6.972	11.806***	2.235	5.638**	2.511
Model performance								
Observations	2,320		2,390		2,450		2,440	
N	232		239		245		240	
AIC	2,964		2,956		3,574		3,394	
Pseudo *R*-squared (adjusted)	0.42		0.44		0.34		0.37	
Log likelihood	−1,475		-1,471		−1780		−1,690	

*Statistical significance: * 10%; ** 5%; *** 1%.*

0.014). The percentage of people whose $\mathrm{Time}_{\mathrm{IVoT}}$ is below or equal to their individual-specific earnings is 67% in the rural site and 57% in the urban site. In comparing the results of $\mathrm{Time}_{\mathrm{IVoT}}$ to people's individual-specific earnings we find that at the median people's $\mathrm{Time}_{\mathrm{IVoT}}$ equals 74% of their earnings in the rural site and at 83% of their earnings in the urban site. The percentage of respondents

TABLE 5 | Results of the RPL models for both the rural and urban converted time experiments

Attribute	Rural experiments						Urban experiments					
	Time payment vehicle: minimum wage		Time payment vehicle: sample earnings		Time payment vehicle: individual value of time		Time payment vehicle: minimum wage		Time payment vehicle: sample earnings		Time payment vehicle: individual value of time	
	Coefficient	SE	Coefficient	SE	Coefficient	SE	Coefficient	SE	Coefficient	SE	Coefficient	SE
Means of random parameters												
Erosion control	−2.626***	0.125	−2.626***	0.125	−2.653***	0.126	−0.352***	0.061	−0.352***	0.061	−0.333***	0.060
Fish abundance	0.026***	0.004	0.026***	0.004	0.025***	0.004	0.055***	0.004	0.055***	0.004	0.053***	0.004
Visitors	0.006**	0.003	0.006**	0.003	0.006**	0.003	0.007***	0.002	0.007***	0.002	0.006***	0.002
Time payments	−4.358***	0.126	−5.908***	0.127	−5.973***	0.121	−4.783***	0.169	−6.511***	0.170	−6.669***	0.212
ASC opt-out	−32.394***	8.607	−34.652***	9.388	−34.363***	9.493	−6.988*	4.001	−6.985*	3.999	−7.430*	4.416
Standard deviations of random parameters												
Erosion control	2.626***	0.125	2.626***	0.125	2.653***	0.126	0.352***	0.061	0.352***	0.061	0.333***	0.060
Fish abundance	0.026***	0.004	0.026***	0.004	0.025***	0.004	0.056]5***	0.004	0.055***	0.004	0.053***	0.004
Visitors	0.030***	0.010	0.030***	0.010	0.029***	0.010	0.030***	0.007	0.030***	0.007	0.030***	0.007
Time payments	0.0	--	0.0	--	0.0	--	0.0	--	0.0	--	0.0	--
ASC opt-out	37.214***	9.228	39.381***	9.995	38.952***	10.076	5.321	5.059	5.320	5.057	5.934	5.461
Model performance												
Observations	1860		1860		1860		1930		1930		1930	
N	186		186		186		193		193		193	
AIC	2,169		2,169		2,164		2,626		2,626		2,634	
Pseudo *R*-squared (adjusted)	0.47		0.47		0.47		0.38		0.38		0.38	
Log likelihood	−1,077		−1,077		−1,074		−1,306		−1,306		−1,310	

*Statistical significance: *10%; **5%; ***1%.*

whose $\mathrm{Time}_{\mathrm{IVoT}}$ is below or equal to the minimum wage, used for $\mathrm{Time}_{\mathrm{MW}}$, is 12% in the rural site and 4% in the urban site.

Model Estimation and WTP Results

The results of the RPL models are presented in **Table 4** and **Table 5**. Positive preferences are identified for all ecosystem services in both study sites. Results of the Krinsky and Robb (1986) simulations are included in **Table 6**. Higher WTP values are found for the time payments for each conversion rate, in both study sites, and for all ecosystem services. We furthermore identify large differences when comparing $\mathrm{WTP}_{\mathrm{money}}$ to $\mathrm{WTP}_{\mathrm{time(MW)}}$ and especially $\mathrm{WTP}_{\mathrm{time(SE)}}$ and $\mathrm{WTP}_{\mathrm{time(IVoT)}}$. The WTP values for the rural visitors attribute are about 46 times larger for $\mathrm{WTP}_{\mathrm{time(SE)}}$ than for $\mathrm{WTP}_{\mathrm{money}}$. The WTP values for the urban erosion attribute are about 20 times larger for $\mathrm{WTP}_{\mathrm{time(IVoT)}}$ than for $\mathrm{WTP}_{\mathrm{money}}$. The identified differences in the distributions across $\mathrm{WTP}_{\mathrm{money}}$ and the time-based WTP results are significant (Mann-Whitney U, p = 0.000, for all attributes in both sites).

In **Figure 6** and **Figure 7** we plotted the WTP distributions of $\mathrm{WTP}_{\mathrm{time(MW)}}$, $\mathrm{WTP}_{\mathrm{time(SE)}}$ and $\mathrm{WTP}_{\mathrm{time(IVoT)}}$ for each of the ecosystem services and per study site. Including all distributions in one figure was not possible since the distributions of $\mathrm{WTP}_{\mathrm{time(MW)}}$, $\mathrm{WTP}_{\mathrm{time(SE)}}$ and $\mathrm{WTP}_{\mathrm{time(IVoT)}}$ are much more dispersed and therefore would not be visible when combined with $\mathrm{WTP}_{\mathrm{time(days)}}$ and $\mathrm{WTP}_{\mathrm{money}}$. The distributions of $\mathrm{WTP}_{\mathrm{time(days)}}$ and $\mathrm{WTP}_{\mathrm{money}}$ are included in separate figures in **Supplemental Appendix C**. The results of Mann-Whitney U tests show that all distributions are significantly different from each other, except for the urban as well as rural distributions of $\mathrm{WTP}_{\mathrm{time(SE)}}$ and $\mathrm{WTP}_{\mathrm{time(IVoT)}}$ for the visitors attribute.

Differences in Market Integration Levels and WTP Disparities Across Study Sites

The results of the analysis on the differences in market integration measures across our study sites is presented in **Table 7**. These results show that the two sites differ

TABLE 6 | Results of the Krinsky and Robb simulations for all rural and urban experiments

Attribute	Rural experiments					Urban experiments				
	Money	$Time_{days}$	$Time_{MW}$	$Time_{SE}$	$Time_{IVoT}$	Money	$Time_{days}$	$Time_{MW}$	$Time_{SE}$	$Time_{IVoT}$
	Mean WTP_{money}	Mean $WTP_{time(days)}$	Mean $WTP_{time(MW)}$	Mean $WTP_{time(SE)}$	Mean $WTP_{time(IVoT)}$	Mean WTP_{money}	Mean $WTP_{time(days)}$	Mean $WTP_{time(MW)}$	Mean $WTP_{time(SE)}$	Mean $WTP_{time(IVoT)}$
Erosion control *Per 1 m reduction in erosion*	28.09***	12.38***	205.14***	966.57***	1,041.28***	12.10***	5.36***	42.05***	236.63***	262.52***
Fish abundance *Per 1% increase in abundance*	0.30***	0.17***	2.02***	9.53***	9.91***	1.57***	0.65***	6.55***	36.88***	42.07***
Visitors *Per 1% increase in visitors*	0.05*	0.03**	0.49**	2.29**	2.46**	0.17**	0.09***	0.82***	4.59***	4.97**

*Statistical significance: *10%; **5%; ***1%*

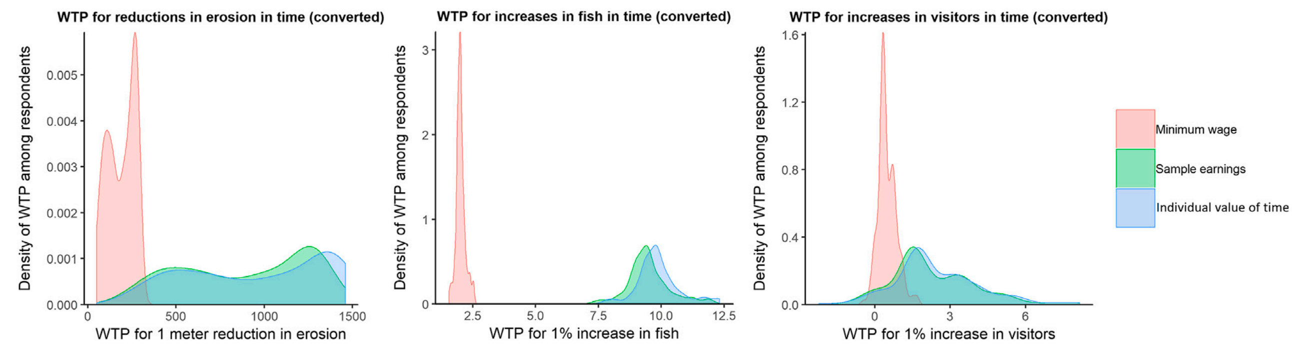

FIGURE 6 | Rural willingness to pay (WTP) distributions for time (converted) payments for the three attributes “erosion control” **(left)**, “fish abundance” **(middle)** and “visitors” **(right)**.

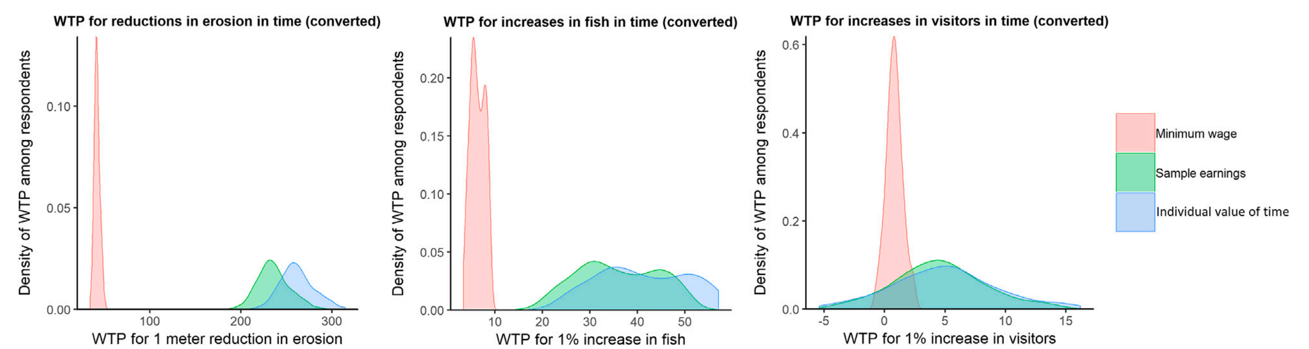

FIGURE 7 | Urban willingness to pay (WTP) distributions for time (converted) payments for the three attributes “erosion control” **(left)**, “fish abundance” **(middle)** and “visitors” **(right)**.

TABLE 7 | Results of the comparison of market integration measures across the urban and rural study site, analysed through Kruskal-Wallis and Chisquare tests

Market integration measure	Measured in …	Urban	Rural	*p*-value
1] Fraction of income from wage labor	% of household income from wage labor	41.79	29.38	0.000
2] Frequency of engagement in wage labor	Days of wage labor per week	2.18	1.74	0.041
3] Income from wage labor	Household income from wage labor in GHC per month	610.52	302.83	0.000
4] Fraction of income from trade profits (not of self-produced goods)	% of household income from trade profits (not of self-produced goods)	19.72	16.67	0.032
5] Frequency of engagement in trade (not of self-produced goods)	Days of trade per week	2.25	1.65	0.004
6] Income from trade profits (not of self-produced goods)	Household income from trade in GHC per month	219.23	139.80	0.000
7] Food that is bought in the household	% of food bought from the store or market	57.87	52.45	0.005
8] Sale of natural resources (self-produced goods)	Number of visits to the market (per year) to sell self-produced goods	43.79	79.30	0.000
9] Likelihood of requesting a loan	Dummy variable for if the household has requested a loan before or not	0.37	0.29	0.016
10] Accumulation of durable goods	Number of durable goods and their market value	4.08	3.35	0.000

TABLE 8 | Results of the analysis procedure to compare urban and rural WTP disparities as described in **Supplemental Appendix B**. Including the results of the Mann-Whitney U tests on whether the size of the WTP disparities differs significantly across the urban and rural study sites, and the median of the percent differences (*PDs*) that represent the WTP disparities between WTP_{money} and WTP_{time} based on the different conversion rates in each site.

Conversion rate	Minimum Wage			Sample Earnings			IVoT		
Attribute	*Median percent difference*		*Mann-Whitney U tests*	*Median percent difference*		*Mann-Whitney U tests*	*Median percent difference*		*Mann-Whitney U tests*
	Urban	*Rural*	*p-value*	*Urban*	*Rural*	*p-value*	*Urban*	*Rural*	*p-value*
Erosion control	347%	726%	0.000	1,953%	3,430%	0.000	2,167%	3,699%	0.000
Fish abundance	416%	673%	0.000	2,352%	3,147%	0.000	2,722%	3,265%	0.000
Visitors	59%	440%	0.000	344%	2,075%	0.000	426%	2,460%	0.000

significantly in terms of market integration levels. We find that people in the urban site participate more frequently in wage labor and trade and also earn more income from these sources, both proportionally and in terms of actual income. Moreover, a higher percentage of the urban respondents has requested a loan before and urban respondents source more of their food from stores and markets rather than from subsistence activities. However, one market integration measure resulted in a significantly higher value for the rural study site: sale of natural resources (measure 8).

In **Table 8** the results of the analysis procedure to statistically compare WTP disparities across the rural and urban study sites, is presented. This procedure is described in detail in **Supplemental Appendix B**. We find that the results of the Mann-Whitney U tests all indicate that there are significant differences in WTP disparities across the study sites. In **Table 8** we also included the median of the *PD*s, which represent the size of the randomly generated WTP disparities for each conversion rate in both the urban and rural study site. Here, we present the median to avoid outlier effects. The results show that for all attributes the *PD*s between WTP_{money} and time-based WTP values are smaller in the urban study site than in the rural study site. For instance, we find that for the erosion control attribute, at the median, $WTP_{time(MW)}$ is about 7.2 times larger than WTP_{money} in the rural study site and about 3.5 times larger than WTP_{money} in the urban study site. **Supplemental Appendix D** includes two additional sensitivity analyses on this matter, discussing differences across samples in terms of market integration measures and whether the full time or converted time samples are used.

DISCUSSION AND CONCLUSION

The aim of this study was to investigate differences in WTP estimates from choice experiments with time and money payments, with the overall goal to estimate more accurate WTP estimates related to NBS benefits. More accurate WTP estimates means higher quality of information on the value of ecosystems and benefits of investing in NBS. Current investment in NBS is low, even though NBS have the potential to contribute to a sustainable and climate resilient future. This is especially the case in developing countries, where the impact of climate change and natural hazards is generally higher. Therefore, we use the results of separate DCEs using time and money payments to value ecosystem services related to NBS in rural and urban Ghana. The NBS include beach nourishment, restoration of coastal lagoons, and recovery of mangrove forest. These NBS are expected to affect the delivery of ecosystem services, including erosion prevention, fish abundance and visitors to the ecosystems. The investigated communities currently face high erosion rates and decreases in fish abundance, both of which are expected to exacerbate with climate change, and thereby increase the need for alternative livelihoods. We first discuss our findings, structured by the four central research questions, and then continue to summarize our conclusions.

Discussion

Generally, our results comply with studies that have found higher WTP values for the time payment vehicle (Alam, 2006; O'Garra, 2009; Casiwan-Launio et al., 2011; Hagedoorn et al., 2020;

Meginnis et al., 2020) but contrast others (Vondolia et al., 2014; Tilahun et al., 2015; Gibson et al., 2016; Navrud and Vondolia, 2020). What is particularly interesting is that our rural study site (Volta delta) is nearby the ones used in Vondolia et al. (2014) and Navrud and Vondoli. (2020). Vondolia et al. (2014) found lower WTP estimates when time was converted using the minimum wage and comparable WTP estimates when time was converted using the sample wage. Navrud and Vondoli. (2019) found lower WTP estimates when time was converted with a wage rate as estimated via a pre-test questionnaire. The main difference between our study and the one from Vondolia et al. (2014) is that their respondents were already used to contributing time as well as money to the management of the irrigation channels. This provides a possible explanation for the difference in results[5]. Details on the type of work that would be conducted by the respondents are lacking in Navrud and Vondoli. (2019), but the study considers different payments for flood insurance that will pay insured farmers after a flood with bags of rice. The difference in results between our study and the one from Navrud and Vondoli. (2019) can potentially also be explained through experience with payments, where it is likely that respondents have more experience with money payments for insurance compared to time payments. More research would be needed to identify how experience with payments for different types of goods affects WTP disparities.

With regard to the size of the WTP disparities, large differences in WTP from time and money experiments have been identified in the literature before (Casiwan-Launio et al., 2011; Hagedoorn et al., 2020). Furthermore, previous studies find that the application of individual-specific conversion rates results in higher WTP estimates compared to generic conversion rates (Tilahun et al., 2015; Hagedoorn et al., 2020). However, the results of both $WTP_{time(SE)}$ and $WTP_{time(IVoT)}$ can be perceived as somewhat extreme, so we put these values into perspective by comparing the WTP estimates for the fisheries attribute to the household income. In the rural study site, mean household income equals 924 GHC and about 25% of this income is related to fishing. In the urban study site mean household income equals 1,280 GHC and about 32% of income is related to fishing. Therefore, mean $WTP_{time(SE)}$ and $WTP_{time(IVoT)}$ values for 1% increase in fisheries of ~10 GHC in the rural site and ~40 GHC in the urban site are not unrealistic, especially since much of the fish catch is used for subsistence and is therefore not included in the households' monetary income. Survey data shows that around 20% of the households' food consumption comes directly from their own fisheries and that between 19 and 25% of the harvested resources never reaches the market due to subsistence use. Nonetheless, on average respondents only work or trade around 3-h per day, which could have led to reduced belief in payment consequentiality in regards to having to work 8-h days. This would also mean that accounting for 8-h of a person's time in $Time_{SE}$ and $Time_{IVoT}$ resulted in an overestimation.

[5]List (2003) finds comparable results in that increased market experience substantially reduces disparities between WTP and WTA value estimates.

Currently, it is common practice in the stated preference literature in developing countries to apply a generic wage-based conversion rate, either the market wage rate or a leisure rate (e.g. Alam, 2006; O'Garra, 2009; Casiwan-Launio et al., 2011; Vondolia et al., 2014; Gibson et al., 2016; Meginnis et al., 2020). Our findings strongly suggest that the underlying implicit assumptions made in using such a generic wage-based conversion rate do not hold. First, we identify high heterogeneity in both wage, trade profit and earnings data and $Time_{IVoT}$, indicating that values of time differ across respondents. Second, we find that there is diversity in the type of activity people would be sacrificing to enable the contribution of time. Therefore, assuming that all respondents would sacrifice the same activity is wrong. Moreover, our study shows that other activities besides wage or leisure may be sacrificed and that different activities hold different values. Third, we find variance in how people's $Time_{IVoT}$ relates to their earnings, suggesting that applying one fraction of the wage rate to obtain a leisure rate (i.e. 1/3 of the wage rate based on Cesario, 1976) to all respondents, does not reflect reality. Additionally, we find that to compose an individual-specific wage-based conversion rate we would have to eliminate more than 30% of our sample or assume that their value of time is equal to 0, which does not comply with our $Time_{IVoT}$ results nor with previous studies' results (Czajkowski et al., 2019; Lloyd-Smith et al., 2019). Therefore, further research on individual-specific non-wage-based conversion rates is recommended.

To elaborate more on this, the results of the non-wage-based conversion rate $Time_{IVoT}$ replicate those of previous studies in the sense of high levels of heterogeneity (Czajkowski et al., 2019; Lloyd-Smith et al., 2019). Furthermore, we also observed weak correlations between $Time_{IVoT}$ and wage and profit data that are within the range of the results of Lloyd-Smith et al. (2019). Compared to Lloyd-Smith et al. (2019), who identified a value of time that averages about 90% of the wage rate, we identify values of time equal to slightly lower fractions of the wage rate, i.e. 74 and 83%. However, for the urban study site we do not detect significant correlations between data on individual-specific wages and $Time_{IVoT}$. Furthermore, we find that the values for the individual-specific wage and trade profit variables are more dispersed compared to the values of $Time_{IVoT}$. These results potentially indicate anomalies in our collected wage data or that part of the respondents answered strategically to the $Time_{IVoT}$ question, potentially stimulated by our consequentiality statement and the involvement of local representatives, enumerators and a local NGO, which could have led respondents to believe that actual paid job opportunities would arise. We were not able to identify nor rule out this behavior based on our data. Therefore, we recommend future studies to include specific consequentiality statements and to work with a more advanced question format for the measurement of $Time_{IVoT}$ as opposed to the open-ended format that we applied. By using an open-ended question, we likely face a lack of incentive compatibility and the measured values of time may present an under- or overestimation (Carson and Groves, 2007). An overestimation seems more likely in our case since income per capita is low in the study sites and therefore respondents may have stated larger values for $Time_{IVoT}$ than that they would be minimally willing to accept as compensation for participating in the restoration work. Applying this method to value time in developing countries could therefore be improved by a different question format, i.e. double-bounded dichotomous choice, if the practical circumstances allow for reliable estimation of time values through such a format. When researchers face a similar situation to ours, i.e. a situation in which it may be complicated to apply different question formats, a simple yet potentially effective qualitative adjustment could come from the inclusion of a provision point mechanism such as described in Bush et al. (2013).

Turning to the market integration measures, the urban study site scores higher on 9 out of 10 of the measures and it therefore seems clear that the urban site is more integrated into the market. However, there is one market integration measure where the rural study site scores higher: the sale of natural resources. Since selling self-produced resources serves as one of the few income opportunities in this area it makes sense that the rural study site scores relatively higher on this measure. As discussed, there is not a uniform way of measuring market integration in the literature. However, most of the relevant literature on WTP disparities from time and money payments focuses on labor and credit market integration as the type of market integration that could reduce these disparities (Casiwan-Launio et al., 2011; Tilahun et al., 2015; Gibson et al., 2016). Therefore, we argue that measures 1 to 6 and 9 and 10 in **Table 8** are more suitable for measuring market integration for this specific purpose. Measure 7 and 8, which are arguably more related to the level of subsistence of the household, might be better suited for measuring market integration in more remote areas such as studies conducted regarding Amazon tribes' market integration.

The combination of the results on differences in market integration across both study sites, and those on differences in the size of the WTP disparities across both sites, provides qualitative evidence on the negative effect of market integration on WTP disparities from time and money payments. This result resonates with previous findings presented in O'Garra (2009) and Hagedoorn et al. (2020) and confirms the theoretical argumentation as laid out in Gibson et al. (2016). We provide a qualitative conclusion on this matter since different respondents answered to different payment types, so we were not able to analyze these effects on an individual level. Moreover, we argue that a household's level of market integration is determined by a combination of the ten market integration measures, which are difficult to combine in one composite variable that could thereafter be added to the RPL models. Therefore, we suggest future studies to further investigate the range of WTP disparities from experiments with time and money payments across settings with varying levels of market integration.

An aspect that has received less attention in our study is the identified heterogeneity in WTP based on time and money payments. We detect heterogeneity in our RPL models, which may arise due to a number of underlying cognitive processes and may have affected our comparison of WTP disparities across study sites. For instance, urban or rural respondents may benefit more from the different ecosystem services and may therefore have stated different WTP, thereby affecting the WTP disparities. Differences in benefits can be related to a respondents' distance from the ecosystems but also to a respondents' occupation, for instance. In both sites, households live close to the coastline (median = 200–400 m in the urban study site and median = 400–600 m in the rural study site) and visit the beaches frequently (median = once a week in both study sites), meaning that the benefits of erosion prevention are likely to be perceived similar across the study sites, although there may still be differences within the study sites. For the fishing attribute, we find that a similar fraction of the respondents is fisher (47% in the urban study site and 56% in the rural study site) as well as fish monger (25% in the urban study site and 29% in the rural study site). For the visitors attribute, where most benefits are likely to end up with the traders, we also identify similar fractions across the study sites (47% in the urban study site and 45% in the rural study site). This means that the effect of heterogeneity on our across study sites comparison is likely to be limited, but within the study sites some respondents may have higher preferences for the fishing attribute whereas others may prefer the visitors attribute. Additionally, heterogeneity in the expected productivity and effectiveness of the contributions may also affect WTP, as do general attitudes towards contributing and more specifically via time or money, and trust in the managers of the contributions. The latter was partly covered by selecting a payment context that receives most trust of the study population as based on the pre-test survey. However, we recommend future studies to investigate the underlying causes of heterogeneity and the effects of this heterogeneity on differences in WTP from time and money payments.

CONCLUSION

In this study we find substantially larger WTP estimates when a time payment vehicle is used compared to when a money payment vehicle is used, thereby answering Research question 1. Furthermore, we answer Research question 2 by showing that the underlying implicit assumptions related to wage-based conversion rates do not hold in the investigated context through a comparison of these conversion rates to a non-wage-based conversion rate. Overall, the identified large WTP disparities from time and money payments likely reflect the problem that financially constrained households cannot freely express their values when monetary payments are required. Simultaneously, it shows that NBS provide highly valued benefits to substantial parts of the population.

Our answer to Research question 3 is that market integration levels are higher in an urban setting than in a rural setting since the urban study site scores higher on all but one market integration measure. This result adds to the identification of larger WTP disparities in the rural study site compared to the urban study site, which allows us to conclude that larger (smaller) WTP disparities correspond to lower (higher) market integration levels, thereby answering Research question 4. However, even though WTP disparities are smaller in the urban study site, we still find significantly higher WTP estimates for the time payment vehicle in this site.

Based on our results we provide two recommendations to those that work on valuing NBS in developing countries. First, we recommend to consider implementing SP studies with time payment vehicles. The two alternative approaches for valuing NBS in developing countries (i.e. SP studies with money payments and the avoided damage cost approach), pose a higher risk of underestimating the values of NBS and may therefore hamper investments in NBS. While considering implementing an SP study with time payments, we recommend to investigate market integration levels during the testing phase of such a valuation study to guide this decision. When low market integration levels are observed, a time payment vehicle is arguably more suitable, and when high levels of market integration are observed, one may want to question the use of a time payment vehicle. Yet, since we still find large WTP disparities in the urban area where market integration levels are higher, we cannot provide an indication of a standard range regarding the level of market integration for which a time payment vehicle would not be suitable anymore. Second, we suggest to convert the estimated time values to money values through a combination of wage-based and non-wage-based conversion approaches and use both results in cost-benefit analyses as a means of sensitivity and uncertainty analysis. On the one hand, we find evidence that the assumptions made in generic wage-based conversion rates do not hold in reality. On the other hand, we also acknowledge that improvements would have to be made to our non-wage-based conversion approach to solve potential issues related to incentive compatibility and strategic bias. These adjustments in the approach to value the benefits of NBS can lead to more accurate welfare estimates of NBS projects, trigger investments and thereby reduce the finance gap that exists for NBS, and contribute to a more sustainable and climate-resilient future.

We also provide recommendations for future lines of research based on the results of this study. First, we recommend to further explore individual-specific non-wage-based conversion rates. We encourage more comparisons of similar and different types of conversion rates as used in this study. This could include similar individual-specific non-wage-based conversion rates or approaches that are based on more advanced question formats, but also individual-specific wage-based conversion rates if the sample characteristics in terms of monetary income sources allows for this. Second, we also encourage studies to further investigate the relation between market integration and WTP disparities. For this purpose, it would be useful to replicate this study in areas with high levels of market integration and measure both WTP disparities and market integration as a kind of benchmark. Subsequently, further replications of our approach in areas with different levels of market integration can contribute to studying the relationship between the level of market integration and the disparities in WTP from time and money experiments. Alternatively, studies may want to pose separate SP questions with time and money payments to one respondent and use this information to analyze the relation between market integration levels and WTP disparities from time and money payments on the individual level. Third, we see potential for further research lines in terms of drivers of WTP disparities in the direction of experience with different payments for different goods, protest behavior related factors such as trust, differences in hypothetical bias across both types of payment, and heterogeneity in benefits received from NBS benefits. Ultimately these endeavors can contribute to discovering the values of natural resources to people in different contexts and different market integration settings.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article is available upon reasonable request from the corresponding author.

ETHICS STATEMENT

Ethical review and approval was not required for the study on human participants in accordance with the local legislation and institutional requirements. The patients/participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

LH, MK and PvB contributed to the conception and design of the study. LH, MK and PvB were responsible for the methodology. LH was responsible for the data collection and analysis. MK and PvB supervised these processes. LH wrote the first draft of the manuscript. LH, MK and PvB contributed to manuscript revision, read, and approved the submitted version.

FUNDING

We are grateful for funding from IUCN-NL and NWO-WOTRO through the Urbanising Deltas of the World programme, project number W07.69.206. We are also grateful for the funds received for the open access publication fee from the Institute for Environmental Studies (IVM) at the Vrije Universiteit Amsterdam.

ACKNOWLEDGMENTS

We also thank our project partners, The Development Institute (Ken Kinney, Senyo Adzah and Tracy Commodore) and University of Ghana (Prof. Kwasi Appeaning Addo, Prof. Barnabas Amisigo and Edem Mahu), for the fruitful collaboration and support in the field. A special thanks to Researchlime (Barnabas Apom) and the 24 enumerators for the hard work and participation in the data collection activities.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2021.686077/full#supplementary-material

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Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Environmental Science

ORIGINAL RESEARCH
published: 26 July 2021
doi: 10.3389/fenvs.2021.678938

Public Acceptance of Nature-Based Solutions for Natural Hazard Risk Reduction: Survey Findings From Three Study Sites in Europe

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OPEN ACCESS

Edited by:
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Reviewed by:
Alexander Van Der Jagt, Wageningen University and Research, Netherlands
Gerd Lupp, Technical University of Munich, Germany

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Specialty section:
This article was submitted to Interdisciplinary Climate Studies, a section of the journal Frontiers in Environmental Science

Received: *10 March 2021*
Accepted: *12 July 2021*
Published: *26 July 2021*

Citation:
Anderson CC, Renaud FG, Hanscomb S, Munro KE, Gonzalez-Ollauri A, Thomson CS, Pouta E, Soini K, Loupis M, Panga D and Stefanopoulou M (2021) Public Acceptance of Nature-Based Solutions for Natural Hazard Risk Reduction: Survey Findings From Three Study Sites in Europe. Front. Environ. Sci. 9:678938. doi: 10.3389/fenvs.2021.678938

Climate change is one factor increasing the risk of hydro-meteorological hazards globally. The use of nature-based solutions (NbS), and more specifically ecosystem-based disaster risk reduction measures (Eco-DRR), has become a popular response for risk reduction that also provides highly-valued co-benefits. Public acceptance is of particular importance for NbS since they often rely on local collaborative implementation, management, and monitoring, as well as long-term protection against competing societal interests. Although public engagement is a common goal of NbS projects, it is rarely carried out with a sufficient understanding of the (de)motivating factors tied to public perceptions. Successful collaboration demands consideration of societal attitudes and values in relation to risk, nature, and place. However, existing research does not sufficiently explore these themes together, their interactions, and their implications for the public acceptance of NbS. This may lead to misaligned public expectations and failed participatory initiatives, while jeopardizing the success of NbS projects and their continued funding and uptake. We conducted citizen surveys within local NbS "host" communities to determine the degree of pro-NbS attitudes and behavior, associated variables, and how these may be leveraged to increase acceptance. We compared results across sites, relying primarily on correlations and regression models along with survey comments and expert knowledge. Three distinct rural NbS being implemented within the OPERANDUM project aim to reduce risk from (socio-)natural hazards in Scotland (landslides and coastal erosion; n = 66 respondents), Finland (eutrophication and algal blooms; n = 204) and Greece (river flooding and water scarcity; n = 84). Our research thus centers on rural NbS for risk reduction within a large EU project. *Trust in implementers* is a consistent factor for defining attitudes towards the NbS across the sites, and attitudes are strongly associated with respondents' *commitment to nature* and *behavioral acceptance* (i.e., willingness to engage). *Behaviorial acceptance* is most consistently predicted by *connectedness to place* and the extent of expected *future impacts*. Skepticism of NbS effectiveness leads to high public demand for relevant evidence. To increase public acceptance, we recommend greater framing of NbS in relation to place-based values as well as demonstration of the effectiveness of NbS for risk reduction. However, distinct hazard types, proposed NbS, and historical characteristics must be considered for developing strategies aimed at increasing acceptance. An understanding of these characteristics and their interactions leads to evidence-based recommendations for our study sites and for successful NbS deployment in Europe and beyond.

Keywords: nature-based solutions (nbs), climate change, public acceptance, public perception, stakeholder engagement, hydro-meteorological hazards, community action

INTRODUCTION

Public attitudes and behaviors are central to tackling the greatest social and environmental issues of our time (Reid et al., 2010; World Bank 2015). The importance of public attitudes and meaningful participation has long been recognized for environmental protection (Blake 1999; Reed 2008) and within the broader context of sustainable development (Chambers 1994). Over the past several decades, the field of disaster risk reduction has undergone a learning process and generally taken up these calls for increasing local and community involvement (Maskrey 1989; La Tozier de Poterie and Baudoin, 2015; Macherera and Chimbari 2016; Begg et al., 2018), spurred on by an understanding of interconnections among environmental protection, sustainable development, disaster risk, and climate change (Turner et al., 2003; Birkmann and Teichman 2010; United Nations 2015; United Nations Office for Disaster Risk Reduction 2015).

Phrases such as “integration of local stakeholder knowledge,” “bottom-up approach,” and any number of verbs following the prefix “co-,” to describe public actions within risk management projects are commonplace. The ubiquity of this terminology is indicative of the shift towards increased reliance on public support (i.e., non-state actors and individuals) (Mees et al., 2012; Penning-Rowsell and Johnson 2015; Bubeck et al., 2017; Begg et al., 2018; Kuhlicke et al., 2020; Zingraff-Hamed et al., 2020; Puskás et al., 2021) that has also been codified in relevant policy such as the European Water Framework Directive (European Commission 2000). Indeed, this shift has been most prominently manifested in the context of flood risk management in Europe (Begg et al., 2011; Begg et al., 2018; Bark et al., 2021) and promoted as a departure from a “decide, announce, defend” practitioner-public interaction model to an “engage, deliberate, decide” approach (Daly et al., 2015). An increasing reliance on the public for addressing environmental risk has been attributed to, among other reasons, a decline in trust in policy-makers (van der Vegt 2018), a push for increased legitimacy and democratic decision-making, a recognition of improved outcomes (Begg et al., 2018; Zingraff-Hamed et al., 2020), the ability to break gridlock and prevent litigation (Irvin and Stansbury 2004), and the extra burden on disaster risk managers due to climate change and land-use conflict (Wamsler et al., 2019).

However, public acceptance and the expected resulting positive outcomes are uncertain and highly predicated on context (Godschalk et al., 2003; Irvin and Stansbury 2004; Euler and Heldt 2018; Wamsler et al., 2019). Additionally, the success of scientific innovations for sustainable development is often determined by public perceptions rather than scientific consensus (Hopkins et al., 2012). Nature-based solutions (NbS) that aim to reduce risk from natural hazards while also providing a wide range of ecosystem services, or benefits, to people (Cohen-Shacham et al., 2016) can be considered one such innovation. NbS encompass measures for ecosystem-based disaster risk reduction (Eco-DRR) and ecosystem-based adaptation to climate change (EbA) (Cohen-Shacham et al., 2016). We focus on Eco-DRR NbS in this study. The substantial funding for NbS research and its ongoing implementation across Europe is indicative of the increasing political and scientific consensus for these measures (Faivre et al., 2017; Zingraff-Hamed et al., 2020; European Commission 2021).

A greater reliance on local stakeholders for cooperation with NbS during implementation, maintenance, management, and monitoring phases means public acceptance is crucial for their success (Ferreira et al., 2020; Anderson and Renaud 2021; Bark et al., 2021; Puskás et al., 2021). The multi-functionality of NbS entails greater opportunity for stakeholder participation but also greater risk of conflict (Naumann and Kaphengst 2015; Connop et al., 2016; European Commission 2021). Additionally, in the short-term NbS can be less effective than other measures and can require increased long-term protection (e.g., conservation) when faced with competing societal interests within their “host” communities (i.e., the groups of local citizen stakeholders living and interacting with NbS) (Kabisch et al., 2016; Anderson and Renaud 2021). Negative public perceptions are commonly considered a potential barrier to NbS uptake (Connop et al., 2016; Heldt et al., 2016; Raymond et al., 2017; Han and Kuhlicke 2019) and the centrality of local stakeholder engagement is reflected in policy-oriented NbS guidelines (International Union for Conservation of Nature 2020).

Although public participation is a common goal of NbS projects and a prominent feature of relevant guidelines (International Union for Conservation of Nature 2020), it is rare that stakeholder engagement processes are based on a thorough understanding of the motivating and conflicting factors related to public perceptions (Zingraff-Hamed et al., 2020). Research for successful NbS has focused more on its physical implementation rather than local public attitudes and supportive behavior, although a recognition of the latter is increasing (Howgate and Kenyon 2009; Buchecker et al., 2013;

Kabisch et al., 2016; Triyanti et al., 2017; Ferreira et al., 2020). There is also increasing attention on stakeholder preferences within NbS projects, although the focus of these studies generally involves the weighting of criteria for instrumental project outcomes (Giordano et al., 2020; Pugliese et al., 2020; Ruangpan et al., 2020), rather than a broader analysis of relevant perceptions and values. This lack of background social science research on NbS for risk reduction can lead to misaligned expectations (Verbrugge et al., 2017) and communities being blamed for the failure of participatory initiatives (Biswas et al., 2009; Barthélémy and Armani 2015). If facilitated without proper intentions and a rich contextual understanding, local participation may be viewed as performative rather than contributory and lead to both negative perceptions and unsatisfactory project outcomes (Irvin and Stansbury 2004; Begg et al., 2018; Euler and Heldt 2018; Wamsler et al., 2019). In contrast, effective risk or project-related communication and meaningful participation is more likely to be successful with an understanding of individuals' perspectives and values (Moser and Dilling 2011; Simon et al., 2013; Raymond et al., 2017; Brink and Wamsler 2019). Transparent participation and framing of communication can enhance identification of shared goals and improve engagement (Buijs 2009; Moser and Dilling 2011; Simon et al., 2013; Everett et al., 2018), even in contexts of inherently misaligned public-practitioner objectives (Pfadenhauer 2001; Williams et al., 2017).

Perspectives and values vary greatly both across and within the contexts of NbS sites and should be explored on a case-by-case basis but with systematic consideration of relevant variables. In their review of 99 articles related to public acceptance of NbS for disaster risk reduction, Anderson and Renaud (2021) identified the variables found to influence acceptance and their frequency in the literature. The variables were classified as being most relevant to the individual, the society, or the NbS measure itself, and the most frequently cited included perceived benefits and trade-offs, effectiveness of risk reduction, cost, risk perception, place attachment, and trust in the responsible party. Many of the variables can also be classified into the general themes of perceptions of risk, nature, and place, the relevance of which is also suggested by prior research. For example, perceived concern for hazards (Fordham et al., 1991; Ding et al., 2019) or their negative impacts (Böhm and Hans-Rüdiger, 2000; Bubeck et al., 2012; Schernewski et al., 2018; Sjoberg 1999, 2000) are widely cited as potential (context-dependent) motivators of (support for) protective action. Similarly, individuals' "acceptance" or intolerance of risk can determine whether they support risk reduction and its required personal or community resources (e.g., time or money) (Fischhoff et al., 1978; Baird 1986; Chowdhury 2003; Buchecker et al., 2016; Holstead et al., 2017).

Since both using natural elements and supporting ecosystems are central to NbS, the long-standing and well-established research on determinants of pro-environmental attitudes and behaviors is also highly relevant (Liere et al., 1980; Stern 2000; Steg and Vlek 2009). Cleaner air (Groot and Groot 2009; Miller and Montalto 2019) and water (Schaich 2009; Koutrakis et al., 2011) and greater biodiversity (Howgate and Kenyon 2009; Schaich 2009; Jones et al., 2012; Roca and Villares 2012; Scholte et al., 2016; Everett et al., 2018; Miller and Montalto 2019) and wildlife habitat (Kenyon 2007; Herringshaw et al., 2010; Evans et al., 2017; Beery 2018) can be crucial for public acceptance of NbS. The perceived importance of positive environmental outcomes as motivators is related to individuals' sense of interdependence and commitment to nature (Davis et al., 2011).

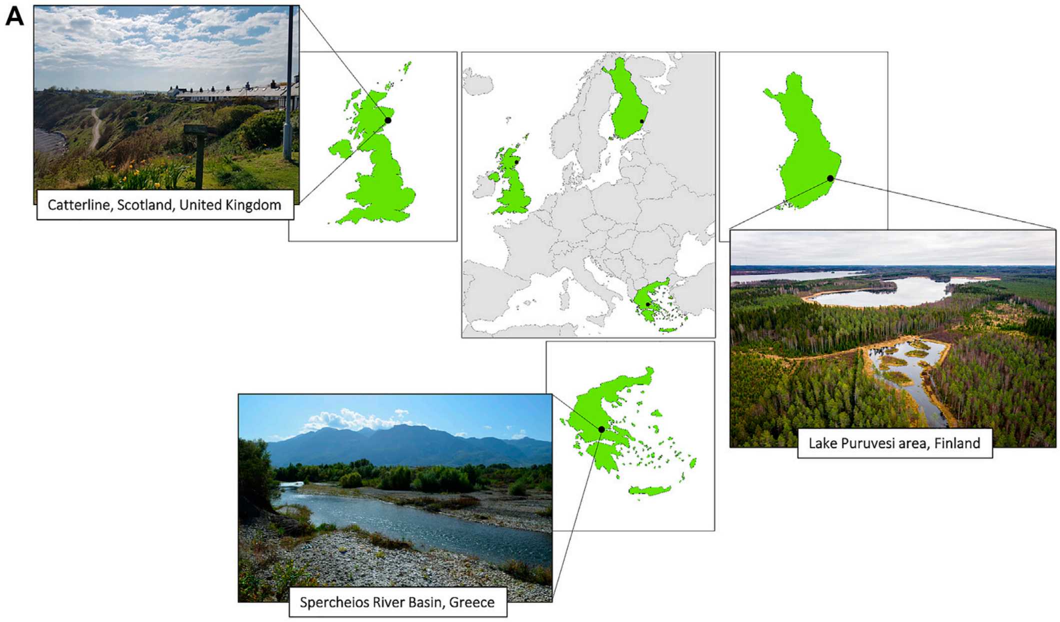
Lastly, whether NbS are seen to enhance or degrade local history, identity, and place can influence the degree of public acceptance (Buijs 2009; Bihari and Ryan 2012; Roca and Villares 2012; Brink and Wamsler 2019). Individuals' connectedness to place may determine whether shifts away from the status quo or the idealized environment face opposition (Buijs 2009; Jacobs and Buijs 2011; Pueyo-Ros et al., 2019) or if NbS that enhance local values find support (Schmidt et al., 2014; Brink and Wamsler 2019). Recent literature reviews on the subject have also found risk, nature, and place to be key themes of variables that influence perceptions of NbS across diverse geographic and hazard contexts (Han and Kuhlicke 2019; Anderson and Renaud 2021).

These research streams from disaster risk reduction and risk perception, environmental attitudes and behavior, and attachment/connectedness to place provide fertile ground for explaining public acceptance of rural NbS (projects) for risk reduction. However, the associated variables from these fields have only very rarely been considered within the same studies on public acceptance (Buijs 2009). Our research addresses the resulting insufficient understanding of what determines public attitudes and behavior in this context.

The ongoing EU-funded OPERANDUM project[1] is implementing NbS in Europe to reduce risk from hydro-meteorological hazards. We conducted surveys with residents of three rural OPERANDUM NbS host communities across Europe (Scotland, Finland and Greece). By 1) assessing public attitudinal and behavioral acceptance of NbS and 2) determining what variables define and are related to acceptance, we aim to address the outlined knowledge gaps and help ensure successful NbS within the study sites while also identifying more general lessons and recommendations for NbS.

We define public acceptance broadly to encompass, for example, cooperation, engagement, satisfaction, and buy-in while avoiding conflict, opposition, and a lack of participation (Anderson and Renaud 2021). It thus describes both attitudes and behaviors toward NbS while recognizing their potentially distinct motivators. We use a comparative research approach to identify similarities and divergence across the sites (Przeworksi and Teune 1970; Lijphart 1975; Mills et al., 2006). Our research compares three rural sites that were all in the mature planning stage prior to deploying NbS for risk reduction. However, within the limits of the OPERANDUM project, our study site selection then seeks to maximize contextual differences across sites in terms of social and environmental systems. This research design can be described as the "most different system" approach (Przeworski and Teune 1970). Divergence in results across sites demands a systematic

[1]https://www.operandum-project.eu



B

Study Site	Study site area	Hazard	Primary NbS	Primary aim of NbS
Catterline	0.4 km^2	Landslide	Live cribwall, live grating, live pole drain	Slope protection and erosion control
Puruvesi	73 km^2	Eutrophication and algal blooms	Continuous cover forestry (CCF)	Erosion control and reduction of runoff
Spercheios	92.4 km^2	River flooding and water scarcity	Natural water retention basins	Excess surface runoff storage and groundwater recharge

FIGURE 1 | Three European NbS study sites **(A)** and their characteristics, including hazard type and primary NbS being implemented within the OPERANDUM project **(B)**. Map: European Commission, Eurostat, https://ec.europa.eu/eurostat/web/gisco/geodata/reference-data/administrative-units-statistical-units/countries. Photo credits: Catterline, Dr Karen Munro; Puruvesi, Pro Puruvesi ry; Spercheios, KKT-ITC S.A.

exploration of contextual characteristics, while similarities across sites leads to cautious inferences regarding generalizability of the independent variables in contexts of rural and externally initiated NbS projects for risk reduction. Based on the risk, nature, and place literature described above, we set out with the hypothesis that these variables will be influential for public acceptance of each NbS across the sites, testing this using maximally different contexts.

We are not aware of any similar studies that compare results across distinct rural study sites with different natural hazards, social and cultural characteristics, and proposed NbS with the primary objective of disaster risk reduction. We combined the comparative approach with psychometric methods since these are suitable for measuring individuals' perceptions through standardized survey items and composite scales (Borsboom 2005) for bivariate and multivariate statistical analyses.

We first provide a brief background on the study sites and detailed description of survey sampling, survey design, and data analysis. Next, results are structured based on the following research questions:

RQ1) What is the degree of public acceptance within the NbS sites and how does this differ across the sites?

RQ2) What variables define attitudinal acceptance, what is their strength within and across sites, and are perceptions of risk, nature and place associated with them?

RQ3) What variables define, correlate with, and explain behavioral acceptance (i.e., willingness to engage), and do attitudes towards NbS moderate their strength?

We then discuss key findings across the sites, their relation to prior research, and corresponding recommendations for increasing public acceptance of NbS within the sites and beyond. This is followed by a reflection on the study's limitations, the direction of further research needed, and a conclusion.

STUDY SITES

Our three European study sites are Catterline, Scotland, United Kingdom; the Lake Puruvesi area in Eastern Finland, and the Spercheios River Basin in Stereá Elláda, Central Greece (**Figure 1**). All three sites are rural and have relatively low-density populations living nearby who are exposed to hydro-meteorological hazards. Additionally, the sites were all at similar points in their project timeline—the NbS had not yet been deployed by the project, but the stakeholder engagement process had begun and NbS planning was at a mature stage. Because contact had already been made with residents during limited prior outreach activities, there was a baseline level of awareness of the OPERANDUM NbS work among the respondents. These three sites were selected within the OPERANDUM project to cover a diverse set of social and environmental contexts, including spatial scales (Catterline is much smaller than the other sites), as well as diverse hazards and NbS. In this way, the survey variables are tested for both their site-specific and general relevance.

Catterline, Scotland, United Kingdom

Catterline is a small, rural, and scenic seaside village in Northeast Scotland with important historic and cultural relevance. The community has a long history of landslides, soil erosion, and related coastal hazards (Gonzalez-Ollauri and Mickovski 2017). Prolonged periods with heavy rainfall, surface water accumulation, fluctuations in groundwater, spring tides, storm surge, and high winds are all long-standing issues that contribute to landslides in Catterline. The last major landslide event, before the surveys were conducted in September 2019, occurred in October 2012.

Along with a detraction from the scenic beauty, the impacts of landslides in the community are most frequently road closures that can inhibit both the residents' recreational opportunities and access to essential services. There is also a fear of property damage or personal injury since past landslides have come within meters of residences.

Recent work to mitigate landslide risk has involved live ground anchor systems and live drainage systems making use of locally available willow branches, as well as the (re)planting of woody seedlings and cuttings along some sections of the slopes. These measures, including live cribwalls and grating, were being planned for deployment by the OPERANDUM project when the surveys were carried out. Additionally, a stabilization effort using geogrid mesh with vegetation was completed in August 2019 by members of a community group—the Catterline Braes Action Group (CBAG[2]). The group was formed following landslides during the winter of 2012/2013. Most members live in the village and it is supported by voluntary resident engagement, with several highly engaged residents and many others supportive.

Lake Puruvesi Area, Finland

Lake Puruvesi and its surroundings in South-eastern Finland are rural, scenic, and culturally significant. Puruvesi is particularly well-known for its water clarity. While most of the 416 km^2 lake is in excellent ecological condition, the frequency of blue-green (cyanobacterial) algal blooms related to eutrophication has increased within portions of the lake, particularly in its north-western extent near the Lake Kuona-Vehkajärvi sub-catchment area.

The dominant land-use in the Lake Puruvesi catchment is forestry (92% of the catchment land area) and the remainder mostly agricultural (7%)[3]. Runoff from rainwater and snowmelt carries sediment and agricultural inputs to the lake. Forestry practices underlie the issue, while the hydro-meteorological conditions for the processes are exacerbated by climate change. Eutrophication occurs when the water is overly enriched with nutrients, often indicated by blue-green algal blooms, lower water clarity, sliming, higher quantity of mud and reeds on the beaches, as well as reduced oxygen levels for plants and fish. Ecological degradation, in turn, impacts recreational activities such as swimming and fishing as well as livelihoods dependent on the water quality of the lake (tourism and fishing). Additionally, adverse health effects can occur, including skin and eye irritation.

The focus of OPERANDUM NbS work in Puruvesi is on continuous cover forestry (CCF), a sustainable resource management practice involving selective timber harvesting to maintain a forest canopy and vegetation density to reduce runoff while also maintaining forest ecosystem structure and habitat. However, other NbS including constructed wetlands, peak flow control structures, sedimentation ponds and pits and surface runoff fields were also being planned at the time of the survey, as communicated to respondents.

Spercheios River Basin, Greece

The steep slopes of the Spercheios River Basin, present within approximately two-thirds of the total length of the river, form a mountainous topography with relatively high flooding peaks and very intense sediment yield. In the last downstream part of the Spercheios course, the topography gradually changes into a lowland relief, discharging into the Maliakos Gulf connected to the Aegean Sea. Our research concentrates on the mouth of the Spercheios River near the city of Lamia, the area with the largest population exposed to flooding. Topography, soil properties and climate are conducive to seasonal flash-flooding and high sedimentation. Along with some tourism, agriculture is the most common livelihood in the area.

Flood events occur on an almost yearly basis that damage property—both residential and agricultural—and can block access roads. Most recently, flash flooding in 2018 caused extensive damage and disruption for several weeks. Tourism and agricultural livelihoods are thereby affected in addition to transportation and recreation. There are no recorded deaths from flooding.

A system of canals and trenches, most of which have been in place since the 1950s, are the primary flood protection measures in the basin. Berms are also in place to provide protection near settlements. These measures have been maintained and extended in the past decades with varying degrees of (mostly limited) success.

NbS in Spercheios are natural water retention measures (NWRM). Drainage basins using natural materials are being implemented to reduce the risk of flooding by absorbing excess water while also providing wildlife habitat and contributing to groundwater recharge and irrigation needs. In parallel, measures such as dam height reduction and the removal of some longitudinal barriers are being taken to increase river connectivity and support downstream wetlands.

METHODS

Survey Sampling

Self-administered surveys of residents living near NbS deployment sites in the OPERANDUM project were conducted between September 2019 and April 2020. The

[2]https://www.cbag.org.uk

[3]https://www.syke.fi/en-US/Open_information/Spatial_datasets

TABLE 1 | Characteristics of the data collection process and outcomes for each of the three study sites.

Study site	Survey date	Format	Collection method	Detailed description	Response rate (%)	Survey count	Survey count after pre-processing
Catterline	September 2019	Paper-based	Door-to-door	Seventy-two residences were included in the study area and contacted by the lead author, first with a survey notification letter one week prior to visiting the community. The lead author went door-to-door to every residence and all over 18-year-old residents were invited to complete the survey. Surveys were left with residents to be self-administered and collected within several days at the respondents' convenience. Surveys were completed at 60 residences	47.2[a]	67	66
Puruvesi	March-April 2020	Online (eHarava[b])	Postcard with online survey link	First, all 1,662 households within the most affected postal code area (also where the NbS are planned) were contacted with a postcard describing the NbS work and inviting participation in the survey through a URL link. Next, 900 members of a local action group of lake users, ProPuruvesi, were also sent a survey notification email with invitation (an estimated 20% of whom were already contacted through the postcard). A short article in a free local newspaper was published in March 2020 that introduced the project and the NbS as well as informing/reminding readers of the ongoing survey	10.3	228	205
Spercheios	October 2019-January 2020	Paper-based	Focus group, convenience	First, surveys were distributed at the end of a public outreach focus group organized within the context of the OPERANDUM project in the town of Kompotades in October 2019. Thirty surveys were collected from the focus group, to which all surrounding residents were invited. In November 2019, 70 additional paper or electronic versions of the survey were distributed to residents by project partners representing the municipality of Lamia using existing institutional mailing lists and contacts	79	85	84

[a]*Based on Scottish Census (2011) output area S00091368; https://www.scotlandscensus.gov.uk/ods-web/area.html.*
[b]*www.eharava.fi.*

Covid-19 pandemic had not yet affected the study areas at the time of data collection. Ethical clearance for data collection was granted by a dedicated review board at the University of Glasgow and all responses were voluntary and treated anonymously. Due to time and financial constraints, the sampling approaches in the three sites were non-random and aimed to maximize the number of responses rather than ensure representative samples. Due to different contexts and capacities of local collaborators, this meant data collection methods across the sites were distinct (**Table 1**).

The samples included mostly even distributions of gender in Catterline and Spercheios and about 60% more males than females in Puruvesi. The sample in Puruvesi was also older, while the sample in Spercheios was younger than the other sites (**Figure 2**).

Measured Variables

We used exploratory factor analyses (EFA) to derive measured variables for *attitudinal acceptance*, *behavioral acceptance*, and variables related to the themes of risk, nature, and place.

The survey primarily included 1–9 Likert items and several yes/no items (full surveys are provided in **Supplementary Text S1–S3**). This Likert range was selected since acceptance is a bipolar construct (i.e., rejection is also possible) (Boateng et al., 2018) and to capture more variation in responses past the mid-point response, given evidence of generally high acceptance of the measures in the sites through past outreach. We used EFA with promax rotation to interpret oblique factors (Abdi 2003) and generate weighted sum factor scores for scales based on Likert items (Briggs and Cheek 1986; Fabrigar et al., 1999; DiStefano et al., 2009; Boateng et al., 2018) while yes/no responses were coded as 1/0 and summed.

We assessed *attitudinal acceptance* with 13 items due to its lack of established relevant scales and multi-dimensionality based on the five themes of 1) trust in implementers, 2) competing societal interests, 3) sense of personal responsibility, 4) perceived effectiveness of NbS, and 5) acceptance of NbS cost. We also included two general items related to whether the NbS is perceived as "good" and whether the respondent is "satisfied

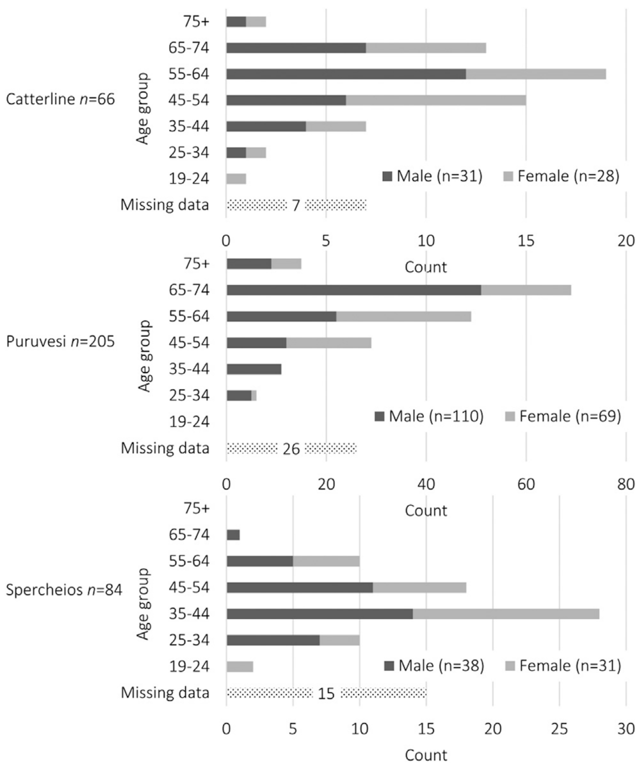


FIGURE 2 | Distribution of male and female respondents in the three study sites by age group. Respondents were explicitly instructed to skip this survey item if they preferred not to respond.

with (ongoing) implementation." The *attitudinal acceptance* themes and all risk, nature and place variables were drawn from the Anderson and Renaud (2021) review on public acceptance of NbS. Variables were selected that are 1) the most frequently cited as influencing public acceptance of NbS, 2) broadly relevant for distinct NbS contexts, including our study sites, and 3) can be assessed using nonintrusive citizen surveys (e.g., do not test for or require extensive NbS knowledge). Additionally, we relied on consultation with local project managers at each of the study sites to ensure the relevance of the variables.

For *behavioral acceptance*, we used six items to reflect the most relevant forms of both passive and active engagement in the sites—"*I would like to . . . :* "*learn about NbS,*" "*attend meetings,*" "*implement and maintain,*" "*monitor,*" "*fundraise or source supplies*" (not applicable in Spercheios), or "*volunteer in other ways*". These items were designed to capture the wide range of potential forms of acceptance identified in Anderson and Renaud (2021). They were determined in consultation with local project managers to 1) include the full range of past supportive actions of residents, 2) include potential future actions that would be instrumentally useful for the project managers (i.e., more than merely performative), 3) be relevant across NbS contexts (i.e., not overly specific to the sites), and 4) capture a range of knowledge, skill, and physical capacities of residents. This latter criterion was particularly important given the substantial elderly population in the Catterline and Puruvesi sites.

We use scales, i.e., internally reliable compositions of multiple survey items that measure a single concept (Borsboom 2005), for *attitudinal acceptance* (13 items) and *behavioral acceptance* (6), as well as for variables within the themes of risk, nature, or place. These include: *risk perception* (5), *risk intolerance* (4–6), *past impacts* (5–8), *future impacts* (5–8), *commitment to nature* (4), and *connectedness to place* 4) (**Table 2**). Risk scales (excluding *risk perception*) vary in number of items due to the number of relevant hazard impacts identified per site (**Supplementary Text S1–S3**). To capture the environmental aspect of an item related to attitudinal acceptance, "*sense of responsibility for risk reduction,*" we included the additional single item—responsibility for nature (Blake 1999). We use the term "variables" to refer to all survey items and scales, with the exception of EFA results for *attitudinal acceptance*, which we refer to as "factors".

The commitment to nature scale is based on Davis et al's. (2011) commitment to the environment scale and the connectedness to place scale on Jorgensen and Stedman (2001). These were truncated due to space constraints (Buijs 2009) and to prevent respondent fatigue and/or criticism of seemingly irrelevant survey material. Risk perception scales relevant to natural hazards in academic literature have historically focused primarily on hazard characteristics (Fischhoff et al., 1978; Slovic et al., 1985; Siegrist and Árvai 2020). Perceived vulnerability and concern (or "worry") have also been associated with risk perception and protective behavior and engagement (Rundmo 2002; Peters et al., 2006; Gifford and Comeau 2011; Terpstra 2011). We combined items related to perceived hazard, vulnerability, and concern and created additional scales of summed binary past impacts (experienced) and future impacts (expected). The risk intolerance scale was inspired by Finlay and Fell (1997), who applied the concept to individual perception of landslide risk, Maynard et al. (1976), who assessed acceptability of risks associated with nuclear waste disposal, and Haynes et al. (2008), who assessed tolerability of volcanic risk.

Generally, the scales yielded appropriate alpha scores. Truncating the scales decreased their reliability and necessitated, in some cases, the iterative exclusion of items on a site-by-site basis (see **Supplementary Table S1** for a list of retained/excluded variables per site) (Boateng et al., 2018). The risk perception scale showed the lowest reliability scores. Due to several low scores, we conducted a final analysis using all underlying single items in addition to the survey scales.

Space was provided periodically for respondents to write in "survey comments," which we assessed to help interpret the results. Translations were carried out by the authors.

Data Pre-Processing and Analysis

Data pre-processing was carried out using Excel and analysis carried out using SPSS (v. 26). Responses with high missing data counts ($n = 14$ in Puruvesi) or with lack of expressed consent were removed ($n = 5$ in Puruvesi; $n = 1$ in Spercheios). Due to small sample sizes in Catterline and Spercheios, single missing values for scale items were imputed using the median of other items for the same scale and respondent (Bernaards and Sijtsma 2000).

TABLE 2 | Composition and computation of variable scales. For scales composed of 1–9 Likert items, processing and reliability testing was conducted by assessing Cronbach's alpha (α), corrected-item-total correlations (CITC), and exploratory factor analysis (EFA) using principal axis factoring. The "original" Cronbach's α is a measure of the internal reliability of all scale items per site (C=Catterline, P=Puruvesi, and S=Spercheios), while the "final" Cronbach's α results from removing items from the scales to increase their reliability, based on the processing steps described. Factor scores using weighted averages were calculated for further analysis.

Scales[a]	Risk perception	Risk intolerance	Past impacts	Future impacts	Commitment to nature	Responsibility for nature	Connectedness to place
Item count	5	4–6	5–8	5–8	4	1	4
Agg. method	Factor score	Factor score	Sum	Sum	Factor score	N/A	Factor score
Themes/item structure	Coping capacity Susceptibility Hazard frequency Hazard magnitude Concern	*"It is okay if [exposed element] is/are affected by [hazard] once every [time span]."*	*"In the past, [hazard] has affected my [exposed element] in [place]."*	*"In the future, I believe [hazard] will affect my [exposed element] in [place]."*	Well-being Attachment Feel good Best interests	*"As a resident of [place], I feel responsible for protecting its natural environment."*	Identity Attachment Dependence Pride
Original Cronbach's α	C = 0.491 P = 0.630 S = 0.576	C = 0.864 P = 0.854 S = 0.851	N/A	N/A	C = 0.887 P = 0.587 S = 0.564	N/A	C = 0.734 P = 0.668 S = 0.724
Final Cronbach's α	C = 0.550 P = 0.653 S = 0.728	C = 0.864 P = 0.854 S = 0.839	N/A	N/A	C = 0.887 P = 0.759 S = 0.695	N/A	C = 0.771 P = 0.651 S = 0.776
Final% variance explained	C = 69.2 P = 51.1 S = 56.0	C = 72.6 P = 81.2 S = 62.3	N/A	N/A	C = 75.4 P = 68.0 S = 63.1	N/A	C = 72.8 P = 59.5 S = 69.9

Scale processing steps

1. Compute Cronbach's alpha scores, alpha if item deleted and corrected-item-total correlations (CITC)
2. In parallel, run EFA using principal axis factoring (100 iterations max), eigenvalues 1, and promax rotation (100 iterations max)
3. Remove items from each EFA model until the following criteria are met, in this general order of importance: alpha maximized; no CITC <0.3; no communality <0.3; no cross-loading factors, low loadings on all factors, or stand-alone large negative loadings; percent variance maximized; adequate KMO and Bartlett's test
4. Rerun this process iteratively, removing one variable at a time
5. Calculate weighted averages (non-refined factor score method) to use for further analysis

[a]*Responsibility for nature is a single item.*

Additionally, "I don't know" responses, included on Catterline and Spercheios surveys, were treated as mid-point responses (5) on the scales for risk perception, risk intolerance, and attitudinal acceptance. Items with greater than 5% imputed data per site are shown in supplementary material, along with data imputation for binary hazard impact items (**Supplementary Table S2**).

The data analysis process was guided by the three research questions and required defining *attitudinal acceptance* and *behavioral acceptance* and then running correlation and regression analyses to determine their relation to risk, nature, and place variables. The results subsections are organized based on the three research questions and corresponding analyses (**Figure 3**).

First, responses to *attitudinal* and *behavioral acceptance* items were divided by the max Likert response (9 for Catterline and Spercheios, 7 for Puruvesi) and arithmetic means calculated to compare the standardized degree of acceptance within and across the sites. Pairwise Mann-Whitney U tests were used to determine any significant differences in means between sites ($p > 0.05$).

We used exploratory factor analysis (EFA) to determine the items that best define the constructs of *attitudinal* and *behavioral acceptance* towards NbS in the sites (see **Table 2** for EFA methodology; **Supplementary Tables S3, S4** for detailed outputs). We then conducted Spearman's rho (ρ) correlations between both *attitudinal* and *behavioral acceptance* and the survey variables related to risk, nature, and place. The correlation analyses allowed us to explore independent associations between acceptance and individual variables. We only report correlations at significance levels of $p < 0.10$, $p < 0.05$, and $p < 0.01$ to simplify visual interpretation of tables.

Multiple linear regression models, in contrast to the correlations, are affected by interrelations among the variables. These were also created using each of the risk, nature, and place variables per site as well as models including all survey variables per site to explain variance in *behavioral acceptance*. For the latter, we included only predictors with the strongest correlations with *behavioral acceptance*, maximum one predictor per eight observations (Wilson VanVoorhis and Morgan, 2007). We

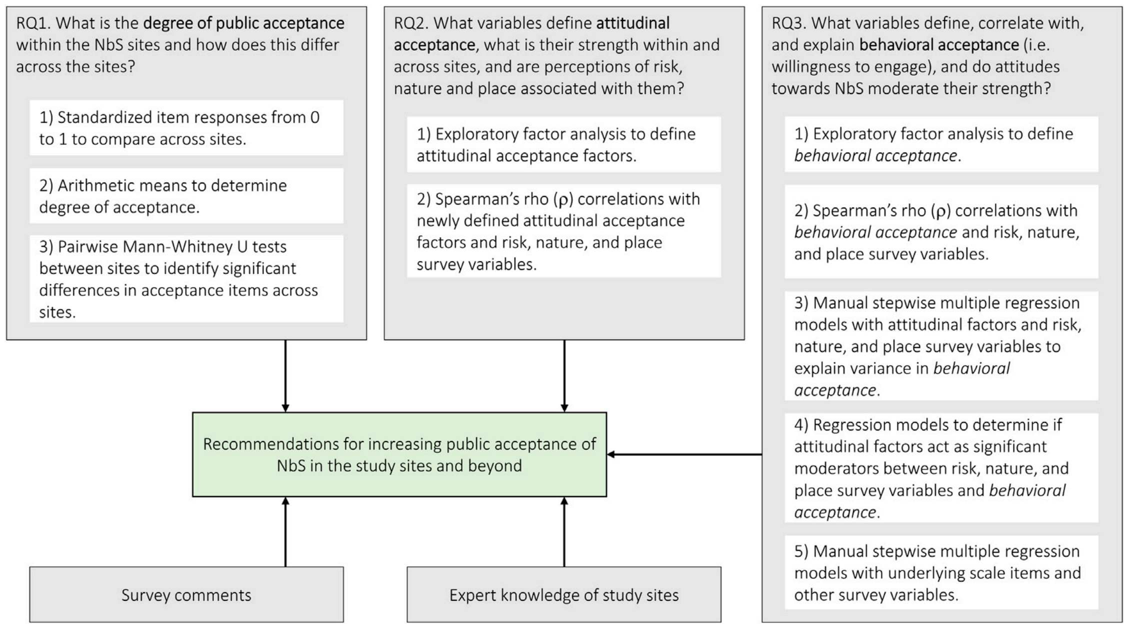


FIGURE 3 | Research questions (RQ) and corresponding methods. Statistical results, combined with survey comments and expert knowledge of the sites, lead to recommendations for increasing public acceptance of NbS both within the study sites and for NbS generally.

followed a manual stepwise procedure of iteratively removing (step-down) the most non-significant predictors until all remaining predictors were significant to at least $p < 0.05$. This is preferred in contrast to relying on automated stepwise regression with biased selection criteria and overemphasis on overall model fit indices (Thompson 1995). It is important to note that excluded predictors are not necessarily insignificant in simple regression models (and therefore relevant) but rather, taken together, do not explain additional variance. Since this method increases the chance of Type I errors within final models, despite all predictor variables grounded in theory as relevant for public acceptance of NbS, we interpreted findings also using correlation outputs, expert knowledge of the sites and qualitative survey comments. The risk, nature, and place variables may be considered underlying personal values and related to affective reactions to NbS, whereas attitudes towards NbS are more analytically driven (i.e., arrived at through reasoning) (Homer and Kahle, 1988; Slovic et al., 2004; Jacobs and Buijs 2011). Therefore, attitudinal acceptance of NbS may moderate the strength of the risk, nature, and place variables on behavioral intention. We created moderating regression models using the PROCESS macro for SPSS (Hayes 2017) with *attitudinal acceptance* factor scores as moderating variables for all risk, nature, and place variables.

RESULTS

RQ1. What is the Degree of Public Acceptance Within and Across the Sites?

Standardized mean responses across the sites show a positive perception of the NbS generally in that their implementation is considered "good" ($M_{Catterline}$ = 0.92/1, SE = 0.022; $M_{Puruvesi}$ = 0.93/1, SE = 0.010; $M_{Spercheios}$ = 0.86/1, SE = 0.023). However, there were lower responses for the degree of satisfaction with how the measures are being implemented ($M_{Catterline}$ = 0.75/1, SE = 0.027; $M_{Puruvesi}$ = 0.79/1, SE = 0.016; $M_{Spercheios}$ = 0.66/1, SE = 0.025) and their perceived effectiveness ($M_{Catterline}$ = 0.71/1, SE = 0.024; $M_{Puruvesi}$ = 0.56/1, SE = 0.024; $M_{Spercheios}$ = 0.72/1, SE = 0.023) (**Figure 4**).

Spercheios stands out as a unique study site among the three regarding acceptance, with nine *attitudinal acceptance* items significantly lower than the other two sites and four *behavioral acceptance* items significantly higher than the other two sites (Mann-Whitney U $p < 0.05$). There is generally greater skepticism surrounding the measures and implementers in Spercheios but more willingness to actively support them. The discrepancy in acceptance values in Spercheios demonstrates the important distinction between the two forms of acceptance.

Greater skepticism in Spercheios is likely in part due to past failed flood protection measures in the region and a mistrust of authorities (Georghiou 1996). This may play a role in the perceived importance of cost as well—a significantly stronger potential limiting variable for acceptance among residents of Spercheios (M = 0.64/1, SE = 0.026; Mann-Whitney U $p < 0.05$) and significantly less of a barrier to acceptance in Puruvesi (M = 0.80/1, SE = 0.017; Mann-Whitney U $p < 0.05$). Two items related to effectiveness, "*NbS will reduce risk in the future*" and "(*do not*) *need more evidence NbS will work,*" have the two lowest average standardized scores across the sites. The other item related to effectiveness describes fatalist or agentic views of the risk, "*risk can be reduced,*" and had the second highest average scores summed across the sites. This indicates that the skepticism regarding effectiveness of risk reduction originates from the

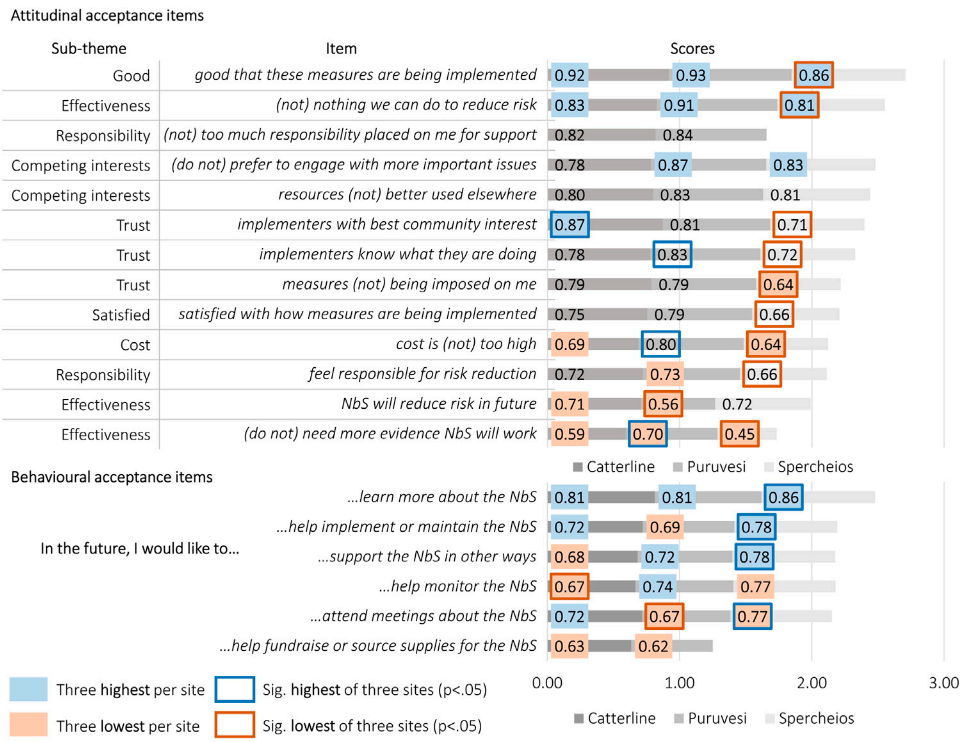


FIGURE 4 | Standardized average responses for survey items that represent *attitudinal acceptance* and *behavioral acceptance* in the three study sites. Response averages are standardized by dividing by the maximum Likert response (9 for Catterline and Spercheios and 7 for Puruvesi), so that 1 is the new maximum value and 0 the minimum. Attitudinal and behavioral items are ordered from top to bottom by highest sum of the average responses across the sites. The top three highest average responses for each site for each form of acceptance are highlighted in blue and the three lowest highlighted in red. Using Mann-Whitney *U* tests, the items that are significantly higher in one site compared to each of the other two sites ($p < 0.05$) are outlined in blue and those significantly lower in red. The third *attitudinal acceptance* item from the top regarding responsibility and the last *behavioral acceptance* item were excluded from surveys conducted in Spercheios since they were not applicable.

specific nature-based solutions rather than from a sense of hopelessness or inevitability.

In both Catterline and Spercheios, there were also a high number of mid-point responses on the Likert item regarding satisfaction in implementation (Catterline n = 14; Spercheios n = 14) and items related to trust, particularly "*implementers know what they are doing*" (Catterline n = 20; Spercheios n = 23). This most likely represents either a lack of information and/or a "wait and see" mind-set, since all NbS were in the pre-implementation phase when the surveys were completed. This mind-set has been prominent in past community outreach activities in Catterline. Items for *behavioral acceptance* show high public demand for both more passive and active forms of engagement with the NbS project. Full descriptive statistics of acceptance items are provided in supplementary material (**Supplementary Table S5**).

RQ2. Attitudinal Acceptance

What Defines Attitudinal Acceptance?

The composition of *attitudinal acceptance* of NbS in the three sites is defined using principal axis factoring. Based on factor loadings, three distinct dimensions of attitudes emerged from the data across the sites. Based on the highest loading factors, we named these: *trust in implementers, benefits outweigh costs*, and *good and satisfied* (**Table 3**).

The factor composition and item loadings are mostly divergent across the sites. It is likely that the unique attributes of each rural NbS site for risk reduction led to differences in the strength of the *attitudinal acceptance* themes and their interrelations. A more comprehensive list of survey items for these themes, and the inclusion of additional themes, may have consistently captured unique dimensions of *attitudinal acceptance*. However, the reasonable percent variance explained (Catterline 62.21%; Puruvesi 73.85%; and Spercheios 79.18%) and the emergence of three unique factors with similarly loading items when considering all three sites suggests that perceptions in relation to trust in implementers, benefits outweigh costs, and good and satisfied with the NbS should be considered when assessing *attitudinal acceptance*.

The first two items related to trust in the implementers were retained together within a factor for all three of the sites. Trust is a particularly large component of acceptance in Spercheios, where the factor composed of these two items explains 53.15% of the variance in attitudes. Different past experiences with flood risk reduction measures and the authorities responsible for them is likely to be crucial here, also supported by the highest standard deviation of scores for these items in Spercheios at SD = 0.24 for each (compared to $SD_{Catterline} = 0.18$; $SD_{Puruvesi} = 0.19, 0.21$). Results suggest that 1) trust towards the implementers of NbS is a unique dimension of acceptance (Spercheios and Puruvesi), and

TABLE 3 | (A) Rotated structure matrix output (promax) from principal axis factoring to determine latent variables of *attitudinal acceptance* in each of the three study sites. Items were standardized for direction when necessary so that increasing scores equated to increasing acceptance. All items were first included and iteratively removed one-by-one from the analysis to maximize reliability and percent variance explained within each site. Two dimensions of *attitudinal acceptance* best explain the variance in each site. Only the higher factor loading between each of the two factors (F1 and F2) per item is shown here, since these were used to derive weighted average factor scores for further analyses. For full scale reliability and EFA outputs (initial and final, after iterative item removal) see supplementary material (**Supplementary Table S3**).

Panel A

		Catterline		**Puruvesi**		**Spercheios**	
	Analysis *n*	66		181		84	
	Total percent variance explained	62.21		73.85		79.18	
	Cronbach's alpha (α)	0.840		0.747		0.704	
	Lowest corrected item-total correlation (CITC)	0.406		0.442		0.427	
	Lowest communality	0.404		0.350		0.501	
	Factor	**F1**	**F2**	**F1**	**F2**	**F1**	**F2**
	Factor percent variance explained	46.03	16.18	50.53	23.32	53.15	26.04
Theme	**Item**						
Good	It is good that these measures are being implemented	0.724					0.707
Satisfied	I am satisfied with how these measures are being implemented	0.761					0.762
Trust	I believe the people implementing the measures know what they are doing	0.657			0.921	0.742	
Trust	I believe the people implementing the measures are doing so in the best interest of the community	0.635			0.741	0.819	
Trust	I (do not) feel that the measures are being imposed on me			0.629			
Competing interests	I believe resources would (not) be better used for other community concerns		0.686	0.909			
Competing interests	I would (not) prefer to engage with more important community issues than (hazard) risk reduction in (place)	0.608					
Effectiveness	I (do not) need more evidence that the natural measures will reduce risk of (hazard)		0.725				
Effectiveness	I believe that when (storms) come in the future, these measures will reduce the chance of (hazard)	0.712					
Cost	I believe the financial cost of these measures is (not) too great		0.667	0.591			

Panel B

	Factor 1	**Factor 2**
Catterline	Good and satisfied	Benefits outweigh costs
Spercheios	Trust in implementers	Good and satisfied
Puruvesi	Benefits outweigh costs	Trust in implementers

2) trust is a consistently important factor for attitudes towards NbS.

Respondents' views regarding competing societal interests, the cost of the NbS, and whether the NbS are "good" and respondents are "satisfied" with their implementation are each retained in two of the three sites. In Puruvesi, perceptions of whether *benefits outweigh costs* explain just over 50% of the variance in attitudes (factor 1; 50.53%). An item *a priori* linked to trust in implementers ("*measures being imposed on me*") loads with the themes competing interests and cost in Puruvesi (0.629), suggesting it is also more related to a cost/benefit judgement of the measures.

Both items designed to capture respondents' sense of responsibility for risk reduction and an item related to an agentic vs. fatalistic view of risk (*a priori* grouped with effectiveness variables; "*risk can be reduced*") were excluded based on low scores for alpha, CITC, and communality (**Supplementary Table S3**).

What Correlates with Attitudinal Acceptance?

Spearman's rho (ρ) correlations show consistently moderate significant correlations of most variables in Catterline and Puruvesi and only *risk intolerance* and *commitment to nature* in Spercheios (**Table 4**). We show here only correlations of at least $p < 0.10$ to ease interpretation of the findings.

Commitment to nature is a significant correlate across all three *attitudinal acceptance* factors and sites. It is particularly associated with respondents' perception of benefits versus costs in Puruvesi ($\rho = 0.518$, $p < 0.01$), as is *responsibility for nature* ($\rho = 0.324$, $p < 0.01$). This is unsurprising since the hazard of eutrophication is itself a degradation of the natural environment. However, the correlation of $\rho = 0.340$ ($p < 0.01$)

TABLE 4 | Spearman's rho (ρ) correlation coefficients of *attitudinal acceptance* factors and risk, nature, and place survey variables in the three study sites. Only correlations significant to at least $p < 0.10$ are shown.

	Good and satisfied		Benefits outweigh costs		Trust in implementers	
	Catterline	**Spercheios**	**Catterline**	**Puruvesi**	**Spercheios**	**Puruvesi**
Risk						
Risk perception	0.345^{a}			0.340^{a}		0.304^{a}
Risk intolerance	0.261^{b}	0.426^{a}		0.193^{a}	0.257^{b}	0.125^{c}
Past impacts (sum)	0.401^{a}			0.142^{c}		
Future impacts (sum)	0.489^{a}			0.212^{a}		0.178^{b}
Nature						
Commitment to nature	0.319^{a}	0.231^{b}	0.229^{c}	0.518^{a}	0.207^{c}	0.301^{a}
Responsibility for nature	0.308^{b}			0.324^{a}		0.179^{b}
Place						
Connectedness to place	0.425^{a}			0.225^{a}		0.240^{a}

[a]*p < 0.01.*
[b]*p < 0.05.*
[c]*p < 0.10.*

with *risk perception* also indicates the intersection between risk and nature in relation to acceptance at the site.

Risk intolerance is also consistently significant and the strongest correlate of any risk, nature, and place variable for Spercheios, associated there with the factor *good and satisfied* at $\rho = 0.426$ ($p < 0.01$). Puruvesi likely has the most significant correlates due to the larger sample size, serving as an important reminder to triangulate correlation results with other statistical outputs as well as expert knowledge and survey comments.

Testing demographic categorical variables of age and gender using simple linear regression, we found that in Puruvesi, gender and age are predictive of positive attitudes in terms of *benefits outweigh costs* ($F (1,181) = 5.75$, $p = 0.018$; $R^2 = 0.031$; $\beta = 0.192$, $p = 0.018$) and gender is also predictive of *trust in implementers* ($F (1,181) = 6.46$, $p = 0.012$; $R^2 = 0.035$; $\beta = 0.186$, $p = 0.012$). There, female respondents have significantly more positive attitudes toward the NbS for *benefits outweigh costs* and for *trust in implementers* (Mann-Whitney U $p < 0.05$) and increasing age predicts increasing positive attitudes of *benefits outweigh costs* ($F (1,181) = 6.39$, $p = 0.012$; $R^2 = 0.034$; $\beta = 0.185$, $p = 0.012$).

RQ3. Behavioral Acceptance

What Defines Behavioral Acceptance?

Based on principal axis factoring, a single factor captures most of the variance in *behavioral acceptance* with high internal reliability in all three sites (Catterline: 75.83% variance explained, Cronbach's $\alpha = 0.933$; Puruvesi: 66.29%, $\alpha = 0.898$; Spercheios: 63.81%, $\alpha = 0.856$). We therefore retained all items and calculated weighted factor scores for further analyses of a single *behavioral acceptance* variable for each site (**Supplementary Table S4**).

What Risk, Nature, and Place Survey Variables Correlate With and Predict Behavioral Acceptance?

Both the *attitudinal acceptance* factors and risk, nature, and place variables are consistently and significantly correlated with behavioral intention across the sites. The attitudinal factor *good and satisfied* has the second strongest correlation of any variable in Catterline ($\rho = 0.492$, $p < 0.01$) and the attitudinal factor *trust in the implementers* has the strongest correlation in Spercheios ($\rho = 0.369$, $p < 0.01$) (**Table 5**).

Although both *risk perception* and *future impacts* are significant correlates across the three sites, the latter is more

TABLE 5 | Spearman's rho (ρ) correlation coefficients of *attitudinal acceptance* factors and risk, nature, and place survey variables with *behavioral acceptance* in the three study sites. Only correlations significant to at least $p < 0.10$ are shown here. N/A (not applicable) is used when the factor did not define *attitudinal acceptance* in that site.

	Behavioral acceptance		
	Catterline	**Puruvesi**	**Spercheios**
Attitudinal acceptance			
Good and satisfied	0.492^{a}	N/A	0.297^{a}
Benefits outweigh costs		0.327^{a}	N/A
Trust in implementers	N/A	0.223^{a}	0.369^{a}
Risk			
Risk perception	0.436^{a}	0.276^{a}	0.252^{b}
Risk intolerance		0.254^{a}	0.264^{b}
Past impacts (sum)		0.319^{a}	0.354^{a}
Future impacts (sum)	0.510^{a}	0.385^{a}	0.286^{a}
Nature			
Commitment to nature	0.324^{a}	0.395^{a}	
Responsibility for nature	0.396^{a}	0.410^{a}	0.219^{b}
Place			
Connectedness to place	0.465^{a}	0.284^{a}	0.330^{a}

[a]*p < 0.01.*
[b]*p < 0.05.*

TABLE 6 | Multiple linear regression model results using attitudinal factor scores and risk, nature, and place variables as initial independent variables and *behavioral acceptance* scores as the dependent variable in each study site. Variables are removed from the model in a step-wise manner in order of least significant *beta* per model, until only *beta* (β) coefficients at p < 0.05 remain.

Model	Predictors	β	R^2	Adj. R^2	*F*	df	DW
Catterline			0.277	0.254	12.09[a]	65	1.85
	Risk perception	0.382[a]					
	Good and satisfied	0.256[b]					
Puruvesi			0.317	0.297	16.22[a]	180	1.92
	Responsibility for nature	0.211[a]					
	Benefits over costs	0.208[a]					
	Future impacts (sum)	0.173[b]					
	Past impacts (sum)	0.162[b]					
	Connectedness to place	0.144[b]					
Spercheios			0.377	0.345	11.76[a]	82	1.71
	Connectedness to place	0.297[a]					
	Past impacts (sum)	0.287[a]					
	Trust in implementers	0.263[a]					
	Good and satisfied	0.241[b]					

[a]p < *.01*.
[b]p < *.05*.

strongly correlated (Catterline ρ = 0.510, p < 0.01; Puruvesi ρ = 0.385, p < 0.01; Spercheios ρ = 0.286, p < 0.01). This is in line with past risk perception research showing that perceived consequences are more associated with mitigative or adaptive behavior than hazard characteristics (Sjoeberg 1999).

In Catterline and Puruvesi, respectively, *behavioral acceptance* (like *attitudinal acceptance*) is shown to be associated with respondents' *commitment to nature* (ρ = 0.324, p < 0.01; ρ = 0.395, p < 0.01) and *responsibility for nature* (ρ = 0.396, p < 0.01; ρ = 0.410, p < 0.01). Landslides in Catterline and eutrophication in Puruvesi are both seen as threats to the ecosystem, in contrast to Spercheios where, despite also being an area of high scenic beauty, the impacts of flooding and drought are felt more in relation to the social system. These results suggest that perceptions of risk *to* nature from the hazards is worth considering for acceptance, in addition to the appreciation of ecosystem services from the NbS.

Connectedness to place is significant across the three sites and particularly strong for Catterline (ρ = 0.465, p < 0.01). A related item on the surveys in Catterline, "*landslides are a threat to our history and culture,*" is also strongly correlated with *behavioral acceptance* at ρ = 0.480 (p < 0.01). In regression models using *attitudinal acceptance* factors and risk, nature, and place variables, *connectedness to place* is one of only three variables retained in two of the sites (along with *good and satisfied* and *past impacts*) (**Table 6**). It is not retained in the Catterline model despite its strong correlation, likely due to also having strong correlations with the remaining predictors (ρ = 0.435, p < 0.01 with *good and satisfied* and ρ = 0.443, p < 0.01 with *risk perception*). The models explain 27.7% (Catterline), 31.7% (Puruvesi), and 37.7% (Spercheios) of the variance in *behavioral acceptance* in each of the three sites and all three models are significant at p < 0.01 (Catterline F (2,65) = 12.09, p = 0.000; R^2 = 0.277; Puruvesi F (5,180) = 16.22, p = 0.000; R^2 = 0.317; Spercheios F (4,82) = 11.76, p = 0.000; R^2 = 0.377).

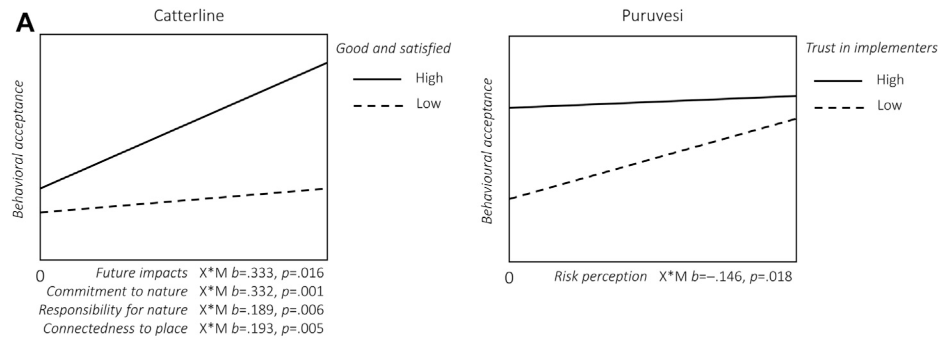
Risk perception is the strongest predictor of *behavioral acceptance* for any of the sites, at β = 0.382 (p < 0.01) in Catterline. In Puruvesi, items related to all three themes of risk, nature and place are significant predictors, as well as the attitudinal factor *benefits over costs*. There, experience of past impacts as well as the perceived potential for future impacts are unique significant predictors (β = 0.162, p < 0.05; β = 0.173, p < 0.05). In Spercheios, attitudes emerge as being particularly important for predicting *behavioral acceptance* (*trust in implementers* β = 0.263, p < 0.01; *good and satisfied* β = 0.241, p < 0.05), along with *past impacts* (β = 0.287, p < 0.01) and *connectedness to place* (β = 0.297, p < 0.01). This finding for Spercheios suggests that strategies aimed at increasing positive attitudes towards the NbS may translate into increased public engagement. Appealing to public pride in place is warranted, a theme returned to in the discussion.

Also noteworthy is the absence of *risk intolerance* from any of the models. Its lack of explanatory ability beyond risk perception and impact scales may be in part due to low variation of skewed right responses for its items (generally risk of listed impacts was not at all tolerated by respondents; see **Supplementary Table S6** for descriptive statistics of risk, nature, and place variables). Using simple linear regression, we found that neither age nor gender is a significant predictor of *behavioral acceptance* in the sites.

Do Attitudes Towards NbS Act as Moderating Variables?

We assessed attitudes as moderating the influence of risk, nature, and place variables on *behavioral acceptance* in each site. After testing for moderation effects of the two attitudinal factors per site, we found one significantly moderating variable (p < 0.05) in Catterline (*good and satisfied*) and one in Puruvesi (*trust in implementers*) (**Figure 5**). In Catterline, the factor *good and satisfied* moderates variables related to all three themes of risk, nature and place—*future impacts*, *commitment to nature*, *responsibility to nature* and *connectedness to place*. As "*good and satisfied*" attitudes towards the NbS increase, each of these variables are significantly more predictive of behavior (full output in **Supplementary Table S7**). This suggests that strategies for increasing behavioral acceptance based on the public's perception of future impacts and relation with nature and place may only be successful if they are also able to improve these attitudes towards the NbS.

In Puruvesi, the attitudinal factor *trust in implementers* significantly reduces the effect of *risk perception* on *behavioral acceptance* [F (3,179) = 8.58, p = 0.000; R^2 = 0.126; X*M b = -0.146, p = 0.018]. Significant relations between risk perception and public trust are well-established, albeit contextual (Slovic 1999; Viklund 2003; Siegrist et al., 2005; Siegrist 2019), but less so as



B

Model	Predictor	Moderating attitudinal factor	R^2	F	P	df1	df2
Catterline_1	*Future Impacts*	*good and satisfied*	.294	8.61	.000	3	62
Catterline_2	*Commitment to nature*	*good and satisfied*	.343	10.80	.000	3	62
Catterline_3	*Responsibility for nature*	*good and satisfied*	.258	6.97	.000	3	60
Catterline_4	*Connectedness to place*	*good and satisfied*	.277	7.92	000	3	62
Puruvesi_1	*Risk perception*	*trust in implementers*	.126	8.58	.000	3	179

FIGURE 5 | Schematic representations of statistically significant ($p < 0.05$) moderating *attitudinal acceptance* factors in Catterline and Puruvesi **(A)** and model statistics **(B)**. These factors (M) moderate relations between the risk, nature, and place predictor survey variables (X) and *behavioral acceptance*. For example, in Puruvesi there is a significant positive relation between *risk perception* and *behavioral acceptance*, but this relation is significantly stronger when respondents' scores on the attitudinal factors *trust in implementers* is low. These are schematic representations of relations. Further statistical output and graphs are provided under **Supplementary Table S7**.

interacting variables for risk management demand and corresponding behavior (Bronfman et al., 2008). One explanation for our finding is that residents who do not perceive the implementing authorities as capable of risk reduction (low trust) are more motivated by perceived risk and a desire to reduce it through engagement with the NbS. This is supported by many survey comments suggesting alternative measures to reduce eutrophication, including: reducing variation in water level, implementing and monitoring wastewater regulation, banning fertilizers, and supporting beaver dams (see full survey comments in **Supplementary Table S8**). The finding suggests that risk framing will not increase acceptance of NbS without parallel gains in trust—both in the implementers and (confidence) in the effectiveness of the NbS (the item "*NbS will reduce risk in the future*" received the lowest standardized average response score in Puruvesi of all *attitudinal acceptance* items at 0.56/1; **Figure 4**).

What Other Survey Variables Predict Behavioral Acceptance?

As expected, when considering all survey variables the regression models increase in explanatory power. An item to assess the perceived social norm of risk intolerance—"*other residents believe risk must be reduced*"—in Spercheios emerges as the strongest predictor for any site at $\beta = 0.487$ ($p < 0.01$) (**Table 7**).

Considering all survey variables as independent variables, multiple regression models explain 51.9% (Catterline), 41.1% (Puruvesi), and 46.7% (Spercheios) of the variance in *behavioral acceptance* in each of the three sites and all three models are significant at $p < 0.01$. (Catterline F (4,48) = 11.86, $p = 0.000$; $R^2 = 0.519$; Puruvesi F (6,181) = 20.33, $p = 0.000$; $R^2 = 0.411$; Spercheios F (2,79) = 33.76, $p = 0.000$; $R^2 = 0.467$).

Both *connectedness to place* ($\beta = 0.281$, $p < 0.05$) and *threat to history and culture* ($\beta = 0.251$, $p < 0.05$) are significant predictors in Catterline. This supports prior findings of individuals' relation to place for acceptance of NbS measures (Buijs 2009; Bihari and Ryan 2012; Roca and Villares 2012; Brink and Wamsler 2019) while also demonstrating that behavioral acceptance can be uniquely motivated by both a connection to place and perceived threat to that connection.

In Catterline, past supportive behavior of NbS was the strongest predictor of intention to support the measures ($\beta = 0.363$, $p < 0.01$). This indicates that targeting residents who have already engaged will likely see the greatest uptake. Perhaps more importantly, having residents actively support the measures in some way may lead to further engagement and foster a sense of responsibility for risk reduction (this had a correlation of $\rho = 0.445$, $p < 0.01$) with behavior).

The importance of perceived cost for *attitudinal acceptance* in Puruvesi was highlighted as also important for *behavioral acceptance* ($\beta = -0.235$, $p < 0.01$), along with past and future impacts ($\beta = 0.151$, $p < 0.05$; $\beta = 0.168$, $p < 0.05$).

DISCUSSION

Shared findings across the sites lead to three key recommendations to increase public acceptance of rural, project based NbS for risk reduction. The recommendations, along with corresponding relevant findings, are first listed below. Strategies and site-specific results related to the key themes are then provided in more detail.

1) Demonstrating the effectiveness of NbS for risk reduction should be prioritized and linked to building trust.

There is skepticism among the public regarding the effectiveness of NbS. Trust in implementers is consistently an important factor for defining attitudes towards NbS and there is a high public willingness to actively engage.

2) The public's sense of place, despite being highly context-dependent, should be considered within NbS projects for their successful uptake.

Public connectedness to place is tied to the importance of the beauty, reputation, history, and culture of the sites and is related to *behavioral acceptance*.

3) In line with the benefits provided by NbS, both perceptions of risk and nature, as well as their interactions, are important for acceptance.

Perceptions of nature are consistently associated with *attitudinal* and *behavioral acceptance* across the sites. Perceived risk and particularly the threat of multiple future impacts is an important predictor of *behavioral acceptance*.

TABLE 7 | Multiple linear regression model results **(A)** and standardised beta (β) coefficients **(B)** using attitudinal factor scores and risk, nature, and place survey variables as independent variables and *behavioral acceptance* scores as the dependent variable in each study site.

Panel A

Model	R^2	Adj. R^2	*F*	df	DW
Catterline	0.519	0.475	11.86[a]	48	1.98
Puruvesi	0.411	0.390	20.33[a]	181	1.94
Spercheios	0.467	0.453	33.764[a]	79	1.57

Panel B

Model	Theme	Predictors	β
Catterline			
	Acceptance	Past acceptance (sum of past actions)	0.363[a]
	Place	Connectedness to place	0.281[b]
	Risk	Future impacts (sum)	0.267[b]
	Risk	"Landslides a threat to history and culture"	0.251[b]
Puruvesi			
	Cost	"Financial cost too great"	−0.235[a]
	Connectedness to place/ Dependence	"Enjoy spending my free time at Puruvesi"	0.209[a]
	Nature	Commitment to nature	0.189[a]
	Responsibility	"Feel responsible for risk reduction"	0.177[a]
	Risk	Future impacts (sum)	0.168[b]
	Risk	Past impacts (sum)	0.151[b]
Spercheios			
	Risk intolerance (social norm)	"Other residents believe risk must be reduced"	0.581[a]
	Connectedness to place/ Identity	"Sense of who I am tied to Spercheios"	0.275[a]

[a]$p < 0.01$.
[b]$p < 0.05$.

Commonalities across the sites suggest that these general recommendations are warranted, while site-specific findings must also be considered for acceptance within the OPERANDUM project and taken up in similar contexts (**Figure 6**).

Key Themes and Recommendations for Increasing Public Acceptance

Effectiveness of NbS and Trust in Implementers for Public Acceptance

We found high public demand for evidence of NbS effectiveness for risk reduction. While most respondents believe risk can be reduced and the NbS will have reasonable success, a range of attitudes between cautious optimism and outright skepticism were expressed. Results suggest that the surveys were conducted at a crucial time in the project lifespan in which most residents have cautious positive perceptions. This presents an opportunity to improve acceptance but also a risk of not fulfilling expectations.

Skepticism of NbS effectiveness is likely to be related to both NbS characteristics and broader context. Potential drivers of hesitant attitudes regarding effectiveness include the complexity and novelty of NbS (Schernewski et al., 2018; Seddon et al., 2020), their effects often being less visible to the public (e.g., rainwater absorption in Catterline and Puruvesi) (Duan et al., 2018; Miller and Montalto 2019), and their duration in implementation with a time lag for effectiveness (e.g., dependence on plant growth) (Kabisch et al., 2016; Shah et al., 2020; Anderson and Renaud 2021). Contextual characteristics such as the history of hazard events in the area, climate change and increasing impacts despite the measures, and zero-risk bias mean that managing NbS expectations is crucial. Therefore, adapting to risk, rather than solely mitigating it, should be a priority of NbS projects and clearly communicated to the public.

Ongoing efforts at collecting evidence of NbS effectiveness are well-positioned to increase public acceptance (Davis and Naumann 2017; Faivre et al., 2018; Chausson et al., 2020). However, perhaps the most powerful way to provide such evidence is through participatory citizen science initiatives in which residents can see for themselves the positive results of the NbS (Holstead et al., 2017)—not just risk reduction but also, e.g., biodiversity gains (Davenport et al., 2010; Pueyo-Ros et al., 2019). Findings show a very high willingness to actively engage in the NbS projects. Resources should be devoted to capacity building and involvement in implementation and monitoring, where appropriate. There is a discrepancy in public willingness to engage and the ability of relevant projects to capitalize on this, particularly for monitoring (Doswald et al., 2014; Puskás et al., 2021).

Although the evidence base for NbS is increasing, there is still substantial work to be done in this regard (Doswald et al., 2014; Kabisch et al., 2016; Davis and Naumann 2017; Chausson et al., 2020). Until NbS are well-established and there exists ample evidence of their contextual effectiveness, trust in implementers as a consistent attitudinal determinant of acceptance will be even more heavily relied on and must be maintained and/or strengthened (Howgate and Kenyon 2009). Trust-building should be a continuous priority, since it can be hard to gain but easy to lose in contexts of risk (Slovic 1999).

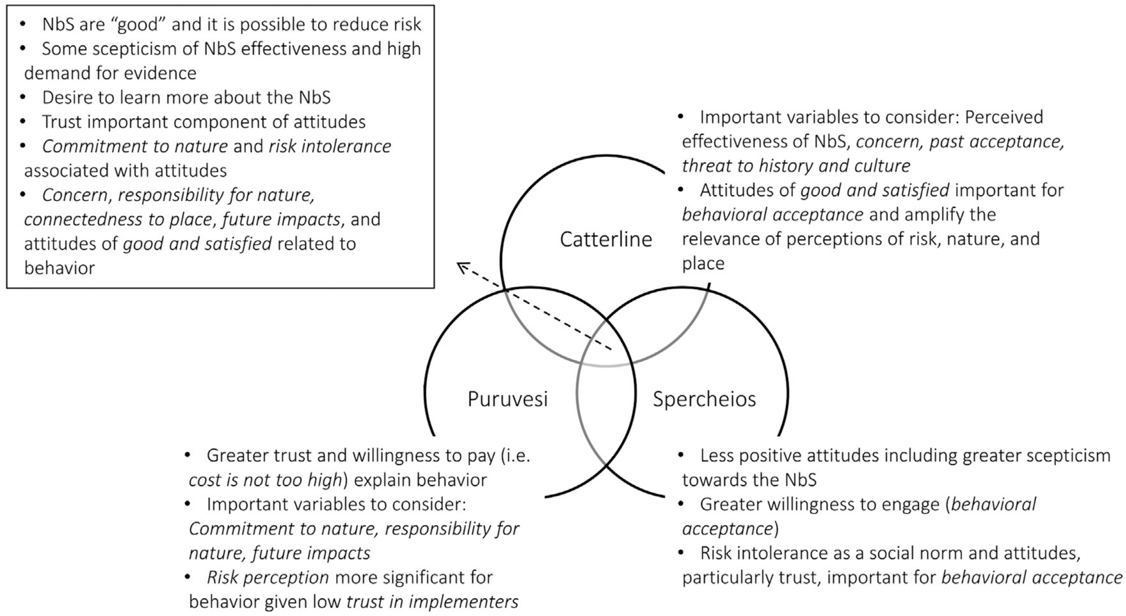


FIGURE 6 | Venn diagram of main findings shared across the study sites and specific findings within each site.

Connectedness to Place for Public Acceptance

In Puruvesi, the strong connectedness to place and the many comments regarding the importance of the reputation of Puruvesi are linked to the NbS and can be leveraged for improving acceptance (**Supplementary Table S8**). As one resident wrote, "*If eutrophication is not controlled, Puruvesi's reputation as Finland's cleanest lake may have been lost. People may also lose hope that eutrophication could be brought under control and stop doing their part for control measures*" (P140). Past research suggests that if NbS are able to enhance highly-valued local natural features, they are more likely to attain public support (Schmidt et al., 2014; Brink and Wamsler 2019). Conveying the importance of the NbS work, not only for improving lake quality but also for the sake of Puruvesi in its context as a highly respected Finnish lake, could be well-received by the public. This act of both localizing the issue and zooming out to the wider implications of NbS efforts will likely be more relevant and motivating for the residents (Buijs 2009; Groot and Groot 2009; Bihari and Ryan 2012; Goeldner-Gianella et al., 2015). Connecting Puruvesi's reputation with eutrophication and its impacts, it may be possible to appeal to the public's pride in- and sense of responsibility for-the natural area.

Connectedness to place was also strongly associated with behavior in Catterline. A related variable, "*perceived threat to history and culture*" was strongly correlated with *behavioral acceptance* and also with the general risk intolerance item of "*risk must be reduced*" ($\rho = 0.566$, $p < 0.05$). This is in line with Buijs (2009), who found that a threat to the perceived historical and cultural setting diminished support for NbS in the context of river restoration. Emphasizing landslides as a threat to place and community, as defined by cultural elements and practices, will likely resonate with residents. For example, amplifying the voices of long-time residents of Catterline in the form of narrative histories of landslide risk in relation to culture could increase knowledge on the issue and promote its position as a communal threat. Crucially, any such efforts must causally link the NbS as an effective actionable solution to the threat to avoid promoting a sense of despair or inevitability (O'Neill and Nicholson-Cole 2009).

Also in Spercheios, connectedness to place was related to acceptance, likely due to regional pride and rural identity. Providing tangible economic benefits in the form of increased tourism or otherwise may improve acceptance of the NbS (Kenyon 2007; Davenport et al., 2010; Roca and Villares 2012). However, this must be approached carefully since not everyone benefits from tourism and a sense of inequity of benefits could be fostered, reducing acceptance (Beery 2018; Otto et al., 2018).

The strongest correlate of *behavioral acceptance* in Spercheios was the item "*other residents believe risk must be reduced*". The perceived social norm of risk reduction is linked to both place and responsibility. Further research should aim at determining whether this is more a function of a moral norm (i.e., "we should act") or a social dilemma (i.e., "I won't act unless others do"), although survey results point more strongly to the former. This finding suggests that strategies for increasing acceptance may be successful by demonstrating that other residents are 1) concerned about natural hazard risk and 2) supporting the NbS work as a result. Testimonials, for example of well-respected and long-standing community members affected by flooding who support the NbS, could be trialed along with publicizing strong attendance at NbS-related activities. Also, pictures of engaged community members or "engagement days" in which locals come out to support the NbS together could be piloted.

Perceptions of Risk and Nature for Public Acceptance

The consistent significant relations between nature-related variables and acceptance reflect the importance of NbS co-benefits and how these measures are framed to the public, i.e., as more than just interventions to reduce risk. One quotation from Catterline captures the recognition of NbS as multi-functional, but primarily intended for risk reduction: "*I think if the measures are as natural as possible this is best for [the] environment and residents. If [they were] manmade prevention methods, I'd be less inclined to support them unless guaranteed benefits*" (C9). Anecdotal evidence from the site also points towards peaks in public engagement in the aftermath of landslides that wanes over time, underscoring temporal fluctuations in the salience of risk and impacts in relation to engagement.

The importance of perceived cost for *attitudinal acceptance* in Puruvesi was highlighted as also important for *behavioral acceptance*, together with the number of past- and future impacts experienced by respondents. Many comments from respondents in Puruvesi reflect varying degrees of perceived severity of the issue of eutrophication (**Supplementary Table S8**), for example: "*...Is there now a fuss about something that can be influenced, when in reality the effect is non-existent?*" (P96); "*Blue-green algal blooms occur in small and predictable areas*" (P193); "*There have hardly been any of them at my cottage beach*" (P205). In this case it seems that the unequal spatial distribution and ephemerality of impacts play an important role in determining whether residents believe the ongoing NbS efforts against eutrophication are worth the resources invested. This is supported by an item for general risk intolerance "*risk must be reduced*" showing a correlation of $\rho = 0.327$ ($p < 0.01$) with the factor *benefits outweigh costs* in Puruvesi. The relatively invisible causal mechanisms behind eutrophication (e.g., rainwater runoff vs. infiltration), may exacerbate this effect. Past research on infrequent hazards and climate change also shows that when threats are perceived as distant in space and time there is less willingness to take action against them (Rambonilaza et al., 2016; Everett et al., 2018; Brink and Wamsler 2019).

The importance of proving the effectiveness of NbS (Miller and Montalto 2019; Chausson et al., 2020), as well as its cost-effectiveness (Davis and Naumann 2017; Faivre et al., 2017; Reguero et al., 2018), is reiterated here. Strategies to demonstrate the negative effects of eutrophication to a greater public than those who are affected by any one algal bloom event are worth considering. Water clarity is a simple and easily relatable indicator of water quality and therefore may be useful for developing persuasive and memorable communication material. It also ties into the importance of the reputation of Puruvesi in Finland as a benchmark for water quality and the strong connectedness to place.

Perceptions of risk are motivators for acceptance and the primary NbS aim of risk reduction should not be detracted from, despite co-benefits being potential additional motivators for NbS acceptance. Nevertheless, natural co-benefits of the NbS are important for increasing acceptance among the wider community and for outreach to residents who may benefit less from risk reduction.

Limitations and Future Outlook

Our survey variables reflect the characteristic of this study as interdisciplinary and exploratory. Many of the variables most strongly related to acceptance are in line with Protection Motivation Theory (Rogers 1975), while the importance of social norms for risk reduction in Spercheios, for example, supports more thorough inclusion of variables and testing also for the Theory of Planned Behavior (Ajzen 1991). Further research should systematically test these theories and others (Kuhlicke et al., 2020) including well-established variables such as self-efficacy for public acceptance of NbS, while also incorporating our findings regarding the importance of nature and place-based perceptions. Although the *behavioral acceptance* scale was highly reliable in all sites (high validity supported by respondents with high acceptance providing their contact information significantly more than respondents with low acceptance; Mann-Whitney U $p < 0.05$), research is also needed to advance scale(s) for assessing attitudinal acceptance of NbS.

Other variables, such as awareness and understanding of the measures, although found to be important for public perception of NbS in recent literature reviews (Han and Kuhlicke 2019; Anderson and Renaud 2021), were excluded from this research. The surveys were self-administered, and we aimed to prevent respondents from feeling "tested" on their knowledge. The OPERANDUM project was ongoing at the time of the surveys and these were carefully designed to not detract from public acceptance by eroding trust or creating stakeholder fatigue. Since our study sites were rural, exposed to hydro-meteorological risk, and the projects externally led, the variables may not apply to other NbS contexts and should be further tested where appropriate. It is possible that connectedness to place is more associated with acceptance where deeply rooted rural identities are prevalent (indeed, segmenting Spercheios data supports this hypothesis) (Buijs 2009). Beyond the internal variables we tested for, research should continue to support the success of NbS through a deeper understanding of the wide range of external considerations (e.g., financial and governmental) (Nesshöver et al., 2017; Wamsler et al., 2019; Seddon et al., 2020), as well as social contexts and issues of practicality that can also determine engagement (Blake 1999). Future public perceptions of NbS depend on their overall success.

We recognize the limitations of our non-randomized single point sampling approach. Additionally, the response rate for Puruvesi was quite low at 10.3%. It is likely that these results show higher acceptance than the population, given that the motivation to complete the survey may represent a certain level of acceptance. However, opposition is also a powerful motivator and it may be that polarized views were over represented, since the written in comments on the surveys also expressed complaints about the NbS work. The broad range of comments and Likert responses bolsters confidence in the surveys having captured more than a specific subsection of the population. Our findings provide baseline evidence for developing strategies to increase public acceptance of NbS. However, all such efforts should first be piloted and segment the public as much as possible. Further segmentation of results presented here are not reported due to space constraints. Our use of multiple statistical tests combined with expert knowledge and survey comments increases confidence in the interpretation and recommendations. However, questions around contextual objectives such as "Should we aim to improve the most negative attitudes towards NbS?" or "Do we need to ensure at least limited public collaboration?" are crucial considerations for further actionable research.

Experiments to test the effects of risk, nature and place framings on acceptance, for example, would help establish causal, rather than just correlate, relations and advance the field (Kuhlicke et al., 2020). Moreover, these designs could overcome the current limitation of assessing behavioral intention rather than actual engagement (Sheeran 2002). The importance of perceptions of nature and benefits versus costs supports the systematic study of perceived ecosystem services of NbS and their relation to public acceptance, including the primary aim of risk reduction (Doswald et al., 2014; Kabisch et al., 2016). Follow-up research to examine these interactions more closely is currently being carried out by the lead authors.

CONCLUSION

Understanding what drives public acceptance of NbS for risk reduction is essential for the success of NbS projects and a first step towards their continued uptake in Europe and beyond. Additionally, public outreach should frame NbS not based on what is assumed to be important to public stakeholders, but rather what is evidenced as being highly valued. Our findings support the importance of perceptions of nature and place in contexts of NbS, along with effective risk reduction.

Despite current support, actively investing in campaigns to improve attitudes and behavior towards NbS rather than assuming continued public acceptance is crucial. Providing benefits through effective NbS is essential, but the burden of proof through evidence is a subsequent hurdle, particularly in the context of increasing risk due to climate change. Our findings not only have immediate practical implications for stakeholder engagement within OPERANDUM study sites but also broader lessons for European and global NbS.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author.

ETHICS STATEMENT

This research was reviewed and approved by the University of Glasgow College of Social Sciences Ethics Committee and follows the GDPR. The participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

Conceptualization, CA and FR; data collection, CA, KM, AG-O, EP, KS, ML, DP, and MS; investigation, CA; data curation, CA; writing—original draft preparation, CA; writing—review and editing, CA, FR, SH, KM, AG-O, CT, EP, KS, ML, and DP; supervision, FR, SH; project administration, FR, KM, AG-O, CT, EP, KS, ML, DP, and MS; funding acquisition, FR. All authors have read and agreed to the published version of the manuscript.

FUNDING

The contribution of CA was supported by a University of Glasgow College of Social Sciences PhD Scholarship. This work was supported by the European Union's (EU) Horizon 2020 research and innovation program. It was funded by and carried out within the framework of the OPERANDUM (OPEn-air laboRAtories for Nature baseD solUtions to Manage hydro-meteo risks) project (Grant No. 776848).

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2021.678938/full#supplementary-material

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frontiers in Environmental Science

CORRECTION
published: 22 September 2021
doi: 10.3389/fenvs.2021.767449

Corrigendum: Public Acceptance of Nature-Based Solutions for Natural Hazard Risk Reduction: Survey Findings From Three Study Sites in Europe

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Keywords: nature-based solutions (nbs), climate change, public acceptance, public perception, stakeholder engagement, hydro-meteorological hazards, community action

OPEN ACCESS

Edited and reviewed by:
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Specialty section:
This article was submitted to Interdisciplinary Climate Studies, a section of the journal Frontiers in Environmental Science

Received: 30 August 2021
Accepted: 03 September 2021
Published: 22 September 2021

Citation:
Anderson CC, Renaud FG, Hanscomb S, Munro KE, Gonzalez-Ollauri A, Thomson CS, Pouta E, Soini K, Loupis M, Panga D and Stefanopoulou M (2021) Corrigendum: Public Acceptance of Nature-Based Solutions for Natural Hazard Risk Reduction: Survey Findings From Three Study Sites in Europe. Front. Environ. Sci. 9:767449. doi: 10.3389/fenvs.2021.767449

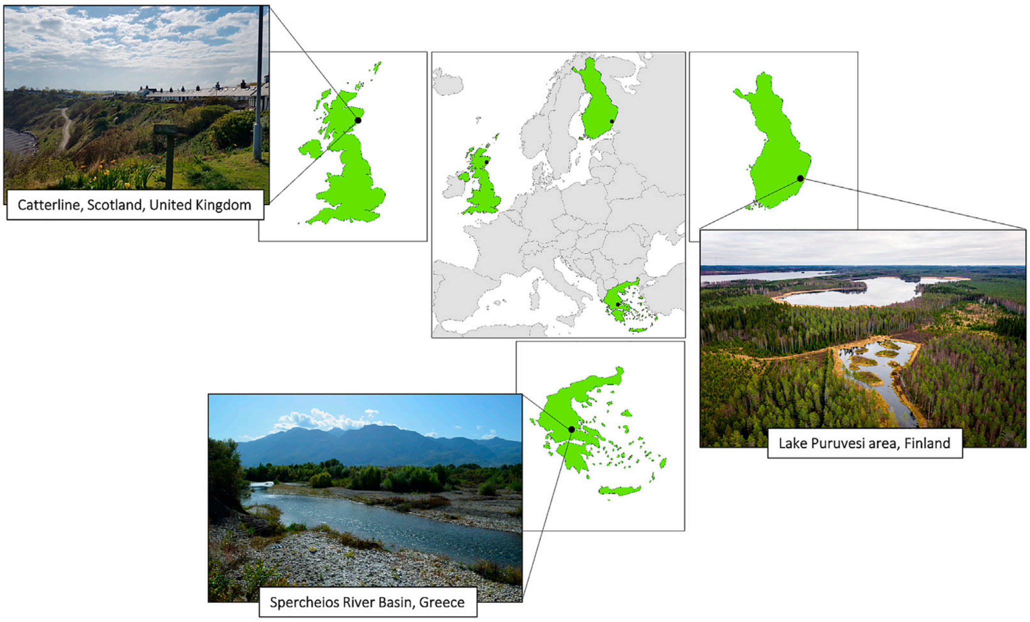
A Corrigendum on

Public Acceptance of Nature-Based Solutions for Natural Hazard Risk Reduction: Survey Findings From Three Study Sites in Europe
by Anderson, C. C., Renaud, F. G., Hanscomb, S., Munro, K. E., Gonzalez-Ollauri, A., Thomson, C. S., Pouta, E., Soini, K., Loupis, M., Panga, D., and Stefanopoulou, M. (2021). Front. Environ. Sci. 9: 678–938. doi: 10.3389/fenvs.2021.678938

In the original article, there was a mistake in **Figure 1** as published. The figure should have panels (A) and (B) but was published only with (A). ***Figure 1B*** *provides characteristics of the NbS study sites, including hazard type and primary NbS being implemented within the OPERANDUM project*. The corrected **Figure 1** appears below.

The authors apologize for this error and state that this does not change the scientific conclusions of the article in any way. The original article has been updated.

A



B

Study Site	Study site area	Hazard	Primary NbS	Primary aim of NbS
Catterline	0.4 km^2	Landslide	Live cribwall, live grating, live pole drain	Slope protection and erosion control
Puruvesi	73 km^2	Eutrophication and algal blooms	Continuous cover forestry (CCF)	Erosion control and reduction of runoff
Spercheios	92.4 km^2	River flooding and water scarcity	Natural water retention basins	Excess surface runoff storage and groundwater recharge

FIGURE 1 | Three European NbS study sites **(A)** and their characteristics, including hazard type and primary NbS being implemented within the OPERANDUM project **(B)**. Map: European Commission, Eurostat, https://ec.europa.eu/eurostat/web/gisco/geodata/reference-data/administrative-units-statistical-units/countries. Photo credits: Catterline, Dr Karen Munro; Puruvesi, Pro Puruvesi ry; Spercheios, KKT-ITC S.A.

frontiers in Environmental Science

POLICY AND PRACTICE REVIEWS
published: 05 August 2021
doi: 10.3389/fenvs.2021.678367

NBS Framework for Agricultural Landscapes

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Entering the UN Decade on Ecosystem Restoration, interventions referred to as nature-based solutions (NBS) are at the forefront of the sustainability discourse. While applied in urban, natural forest or wetland ecosystems, they are underutilized in agricultural landscapes. This paper presents a technical framework to characterise NBS in agricultural systems. NBS in the agriculture sector is proposed as "the use of natural processes or elements to improve ecosystem functions of environments and landscapes affected by agricultural practices, and to enhance livelihoods and other social and cultural functions, over various temporal and spatial scales." The framework emerges from a review of 188 peer-reviewed articles on NBS and green infrastructure published between 2015 and 2019 and three international expert consultations organized in 2019–2020. The framework establishes four essential functions for NBS in agriculture: 1) Sustainable practices — with a focus on production; 2) Green Infrastructure — mainly for engineering purposes such as water and soil, and slope stabilization; 3) Amelioration — for restoration of conditions for plants, water, soil or air and climate change mitigation; and 4) Conservation — focusing on biodiversity and ecosystem connectivity. The framework connects the conventional divide between production and conservation to add functionality, purpose and scale in project design. The review confirmed limited evidence of NBS in agricultural systems particularly in developing country contexts, although specific technologies feature under other labels. Consultations indicated that wider adoption will require a phased approach to generate evidence, while integrating NBS in national and local policies and agricultural development strategies. The paper concludes with recommended actions required to facilitate such processes.

Keywords: green infrastructure (GI), sustainable agriculture, nature-based solutions (nbs), restoration, agrobiodiversity conservation, people-centered, climate-resilience

Edited by:
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Reviewed by:
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Specialty section:
This article was submitted to Interdisciplinary Climate Studies, a section of the journal Frontiers in Environmental Science

Received: *09 March 2021*
Accepted: *20 July 2021*
Published: *05 August 2021*

Citation:
Simelton E, Carew-Reid J, Coulier M, Damen B, Howell J, Pottinger-Glass C, Tran HV and Van Der Meiren M (2021) NBS Framework for Agricultural Landscapes. Front. Environ. Sci. 9:678367. doi: 10.3389/fenvs.2021.678367

INTRODUCTION

Report upon report stress the urgent and pressing state of the world's rapidly degrading natural resources (FAO, 2017; FAO, 2018; FAO, 2019; IPBES, 2019; IPCC, 2019). As we enter the UN decade of Ecosystem Restoration, attention is brought to approaches that integrate natural ecosystems and ecosystems that sustain livelihoods and food production (Sonneveld et al., 2018), conserve or rehabilitate natural ecosystems, and enhance natural processes in modified ecosystems (Cecchi, 2015; Cohen-Shacham et al., 2016; GCA, 2019; IUCN, 2020).

TABLE 1 | Typologies of interventions within NBS for agricultural landscapes, with selected examples relevant for this paper (shaded background). Adapted from Eggermont et al. (2015) to reflect the framework presented in **Table 2**.

NBS for sustainability and multifunctionality of managed agroecosystems	**Design and management of new agroecosystems**	**Better use of natural/protected (agro-) ecosystems**
Partial intervention	**Inclusive intervention**	**None or minimal intervention**
Develops sustainable and multi-functional ecosystems and landscapes that improve delivery of selected ecosystem services; Strongly connected to benefitting from natural systems agriculture and conserving the agroecology	Manages ecosystems in intrusive ways Includes restoration of degraded or polluted areas using grey infrastructures and engineering approached	Maintains/improves delivery of ecosystem services of preserved (agro-)ecosystems; Incorporates areas where people live and work in a sustainable way
Rehabilitation	Rehabilitation	Restoration, conservation
Examples of technological approaches: diverse agroforestry systems, constructed wetlands	Examples of technological approaches: Green infrastructure for slope stabilization, bioremediation, integrated watershed management	Examples of technological approaches: Pollinator flowers, biological pest control, natural regeneration
Field scale to landscape Years to decade	Watershed Decade to decades	Connected landscapes Decades

Beginning in the 2000s, and emerging strongly in development discourses around 2017, nature-based solutions (NBS) gained ground both as a principle and an umbrella of approaches and technologies (Hanson et al., 2020). Deeply rooted in the discourse on ecosystem goods and services (MEA, 2005; Nesshöver et al., 2017), the International Union for Conservation of Nature (IUCN) specified eight principles for NBS (Cohen-Shacham et al., 2016; Cohen-Shacham et al., 2019) which, summarized, embrace nature conservation norms, offer inclusive and context-specific landscape-scale solutions, address societal challenges that produce equitable societal benefits, draw on local and scientific knowledge, address temporal tradeoffs between ecosystem and economic benefits, and are an integral part of policy and regulatory frameworks. As an umbrella, NBS is used to bridge similar concepts and practices for natural and managed ecosystems from different disciplines and for different needs (Cohen-Shacham et al., 2016). For example, the Special Report on Climate Change and Land (IPCC, 2019 p. 739) considers ecosystem-based adaptation (EbA) as "a set of nature-based methods" for adaptation and food security that is closely associated with sustainable land management and water security.

When it comes to specific practices, references to NBS predominantly feature in urban landscapes (Cecchi, 2015; IPBES, 2019) or for conservation and rehabilitation of water and forest ecosystems (Chausson et al., 2020; OECD, 2020; UNDRR, 2020). In particular for urban environments, work has advanced with planning and impact evaluation frameworks for NBS, (Raymond et al., 2017; Albert et al., 2020). Despite its increasing popularity, there is little compiled evidence on the potential of NBS to address problems associated with environmental degradation, disaster and climate vulnerability in agricultural (production) landscapes. For instance, only two out of ten NBS are for agriculture land uses in IUCN's seminal work by Cohen-Shacham et al. (2016). In a systematic approach to categorize interventions for difference ecosystems, Eggermont et al. (2015) practically lay out a typology for how different interventions can maximize the return of ecosystem services from natural and managed ecosystems to the inclusive design of new agroecosystems that can meet the challenges ahead (**Table 1**). This typology provides dynamic benchmarks for many hybrid NBS to enhance their flexibility and problem-solving capacity in agriculture and has been adapted widely. For instance, FAO's framework for NBS in agricultural water management presents a scoring guide for evaluating the success or failure of 21 NBS interventions, where the score represents the degree of ecosystem intervention, benefits of ecosystem services to stakeholders, degree of transdisciplinarity, stability of institutional collaboration, and financing (Sonneveld et al., 2018). This guide focuses on learning lessons and identifying good practice. One benefit of the typology by Eggermont et al. (2015) is it admits the inclusion of the many autonomous 'NBS-like' interventions that smallholder farmers have practiced for centuries, known as local knowledge (Hiwasaki et al., 2014; van Noordwijk et al., 2020), and responds to a multitude of environmental and socioeconomic challenges beyond climate change adaptation (Shah et al., 2019).

Recently, there has been a trend of various "good practices" increasingly being branded as NBS (O'Sullivan et al., 2020), often as compilations of short descriptive cases with elusive criteria for how it qualifies as NBS, good practice, or can be upscaled. For instance, the NBS Coalition of the 2019 UN Climate Action Summit gathered 200 NBS actions for scaling up for mitigation, resilience and adaptation in agriculture, forests, terrestrial and hydrological ecosystems (NBS-Facilitation Team, 2019). Some reports conclude that NBS are flexible, cost-effective and offer multiple solutions (GCA, 2019), and that NBS with safeguards can provide 37% of climate change mitigation until 2030 (IPBES, 2019 p. 10). On the other hand, as corporate and public funding are being availed for NBS, critical voices warn it may mislead as a new "quick fix" as certain "tree-planting" initiatives disqualify as NBS for lack of biodiversity and people-centered considerations (Seddon et al., 2021). Such critique demonstrate challenges with economic valuation of ecosystems (Sonneveld et al., 2018) and also a lacking evidence base on the effectiveness of NBS in the Global South (Chausson et al., 2020; Seddon et al., 2021). Stressing the need for integrated ecosystems approaches adapted for developing country contexts, **Supplementary Box 1** is provided to exemplify the pressing state of agroenvironments across Asia. It also reflects differences and diversity in land use and governance as compared to agroecosystems in Europe and North America, where the theoretical underpinnings of NBS originate.

This paper addresses two of these shortcomings. First, we propose a normative framework for NBS-practices in agriculture, bridging the conventional divide between production and conservation and exemplifying the specific problems NBS offer solutions to. Moreover, we frame NBS as (possibly underutilized) solutions for pressing issues in developing countries. We refer to NBS in the agriculture sector as "the use of natural processes or elements to improve ecosystem functions of environments and landscapes affected by agricultural practices, and to enhance livelihoods and other social and cultural functions, over various temporal and spatial scales." We recognize that the NBS-concept does not substitute nature conservation or conventional "grey" engineering. Instead it offers a way to identify, prioritize and stage solutions that combine traditional, conventional and natural solutions in combinations to generate positive, cumulative biophysical interactions and social benefits.

METHODS

This section outlines the scope of the framework, the iterative process of literature review and the outcomes of expert consultations which helped to refine the framework. The process employed to refine the framework also helped to pinpoint opportunities and barriers for adoption. While developed through interactions with key stakeholders in Asia, we regard it having wider application beyond any one region.

Particularly in developing country contexts, a useful framework should address challenges across a spectrum from production to conservation landscapes, and include land use functions 1) that maintain a high degree of local knowledge and relatively low levels of interventions; 2) for conservation and restoration pathways, and 3) for production systems with various land use management technologies towards restoration and sustainable land uses (**Table 1**). These conditions guided the framework formulation process.

Literature Review and Framework Formulation

The literature search involved three strategies for delineation: 1) Inclusion. NBS feature under different names and concepts, at landscape scale and as practices or technologies within a system (as exemplified in **Supplementary Table 1**). To keep the review manageable, we searched for practices referred to as NBS or GI. This would capture solutions across the spectra of conservation—production agroecosystems as well as engineering based NBS-technologies rarely used for solving problems in agriculture that potentially could lead to innovative land uses or "new agroecosystems" (Eggermont et al., 2015, **Table 1**). 2) Exclusion. Studies from urban contexts or lacking agricultural purpose, having marginal reference to NBS or branded as NBS without theoretical reference were excluded. 3) Screening. Articles remaining after title and abstract screening, underwent full text screening. For remaining articles the reviewers noted name and description of the practice(s) and intention; location, type of landscape and spatial scale; project duration; and for empirical papers, evidence provided for social, economic and ecological benefits. The review team marked references for fulltext review and point to examples 1) implemented in Asia, 2) implemented elsewhere but technically relevant for Asia, 3) providing new insights on economic, social or environmental impacts, or 4) as uncertain for case-by-case exclusion. Empirical studies failing to provide concrete information were ignored. Review articles were used for further references.

The first round of search aimed to establish a working definition and outline the framework. We searched ScienceDirect for abstracts and titles with "green infrastructure" or "nature-based solution." This rendered 3,511 articles, in which the majority referred to urban environments. Adding "'NOT' urban," the result narrowed to 419 articles. After screening, 43 articles remained, and permitted a systematic grouping of practices according to their essential functions and primary purpose. Following the underlying division outlined in **Table 1**, the categories emerged as 1) sustainable practices (must have a productive element), 2) green infrastructure (must have a civil engineering function), 3) Amelioration (must have a beneficial biochemical, biological or microbial function), and 4) Conservation (must have a species preservation benefit), each with two to three qualifying functions.

The second round of search aimed to build more material to work with. We expanded the search to the practices identified as NBS or GI by searching for "practice name X" "AND" ["agriculture" "OR" "fisheries" "OR" "forestry" "OR" "animal husbandry"]. This allowed the inclusion of literature where the practice was not explicitly referred to as NBS or GI. The 25 most recent results in ScienceDirect for each type of practice published between 2005 and 2019 were assessed. In total, 181 out of 1,450 peer-reviewed articles were subject to in-depth review in this step, along with seven of the original 43 articles from the definition stage, a total of 188. This search approach may have excluded practices that are not yet associated with NBS in the literature, although they could have high potential. Furthermore, since the main review was done in 2019, the number of publications on NBS soared. Some updates were made after the consultations in 2020 in preparation for this paper. Making longitudinal or thematic comparisons of search term results was beyond scope in this study. Although the reviewed literature represented some global spread (**Figure 1**), most were based in Europe or North America. Most studies reported for 1) limited spatial (pilot, plot or part of catchment) and temporal scales; 2) 'one' technology rather than sets of NBS-technologies integrated in a landscape or interconnected; 3) one or few monitored environmental indicators, and 4) limited socioeconomic analysis. Similar conclusions were drawn based on the review by Hanson et al. (2020), where 10% of 112 NBS-articles included arable land uses.

Consultations and Further Refinement

The second stage of the framework development involved two phases of consultations. First, the draft framework was presented, tested, and modified at a regional 2-day consultation in Hanoi in July 2019. The 35 stakeholders represented practitioners involved in designing and

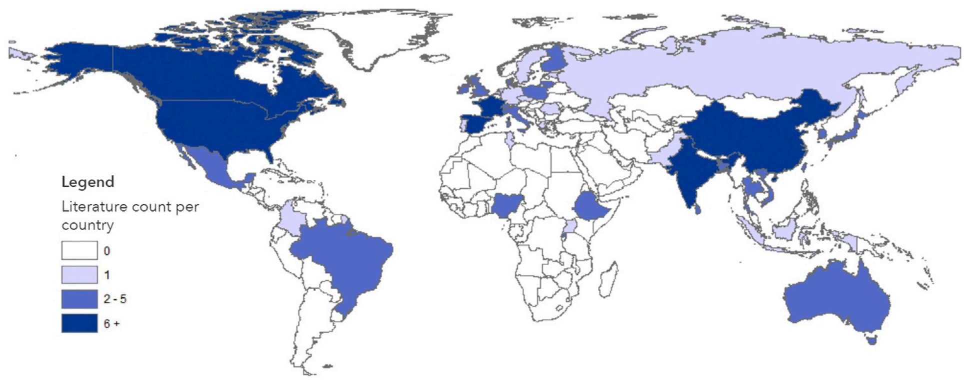


FIGURE 1 | Countries represented in the NBS review (129 out of 188 papers with country-specific field experiments, excluding literature reviews and laboratory experiments).

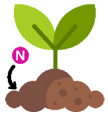
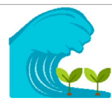
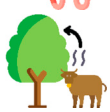
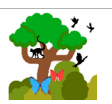
implementing NBS on the ground, national level policy makers, and UN agencies from Indonesia, Lao PDR, Myanmar, Nepal, and Vietnam. The second consultation took place in 2020 with two rounds of online meetings. First with the FAO regional office for Asia-Pacific and its network of country offices followed by a global meeting with the FAO Technical Network on Climate Change, which included representatives from across FAO's technical divisions and global network of regional and country offices.

The consultation rendered important insights for a more problem-based rationale of the framework. First, it is important that any NBS framework be designed with application in mind. In practice, this means that the framework should support the design and implementation of measures (solutions) that will address needs or problems, isolated and cumulative, which result from ongoing and continuous management of agricultural production systems. As an example, one may consider a business-as-usual scenario with conventional farming practices that manages "resource inputs (i.e. fertilizer, irrigation water, amendments, pesticides) uniformly, ignoring the naturally inherent spatial heterogeneity of soil and crop conditions between and within fields [and the] uniform application of inputs results in over and applications of resources" (Corwin and Scudiero, 2019, section 5.8.3).

Related to this, an overarching takeaway from the sessions was the need for human focus in the framework, particularly regarding its utility as a tool for implementation. Both practitioners and policy makers perceived people-centered frameworks to have better chances for implementation and wider uptake than concepts considered as top-down, complex, vague, technocratic, or bureaucratic. These views reflected findings in the review, that NBS-studies from Europe often integrated the general public as beneficiaries of cultural and rural ecosystem services, for recreation and well-being for example. Conversely, studies from developing countries had farmers and local communities as the primary, often only direct users and beneficiaries of ecosystem services, but they rarely interacted in negotiations with the larger society. On the other hand, some argued that too much livelihoods focus risks becoming "another" development project that takes focus away from environmental degradation. Responding to this, two categories were more explicitly integrated into the framework: the spatial and temporal scale of NBS. Incorporating a temporal scale acknowledges that NBS have different effects over different time periods, e.g. short—such as one crop season, medium (1–10 years), or long-term (decades) and has implications for planning of successions of interventions. The full benefits of NBS often emerge on a longer timescale, while unsustainable practices can bring quick short-term gains that hide longer-term negative effects. The spatial scale considers *in-situ* and *ex-situ* impacts. For example, grass strips can have *in-situ* (costs and) benefits for the farmer in the field, and wider *ex-situ* effects, such as amelioration of pollutants or sediments in a river, which will be experienced further downstream. At the largest scale, carbon sequestration measures need to have discernible effects at aggregated scales, up to global. The need for adding scales to the framework also led to the realization of three additional functions: biological pest control and pollination, and, land management practices for the purposes of above and below ground carbon sequestration to the amelioration category.

Discussion also took place at the conceptual level. One question emerged of what NBS could add to EbA, which was considered a more established concept. The question is warranted, as many reviewed papers presented unclear or confounded definitions of NBS and GI (if definitions were present at all). In the agriculture sector, we particularly note inconsistent naming conventions and similar practices and concepts referred to by different names (c.f. **Supplementary Table 1**). Compared to the cases in the Global North, many concepts such as ecosystem services, agroecology, climate-smart agriculture, and NBS have not had the chance to become fully mainstreamed in policy. Therefore, although technically many practices are known, when framed as a new concept it often must undergo a policy integration cycle. Concept fatigue was reflected particularly among the decision makers, who were questioning the need for another concept when policy makers are still struggling to integrate "sustainability" or "ecosystems" in the legislation. A framework with compilations of NBS-practices as in **Supplementary Table 2**, presents a menu of alternatives to and pathways from conventional agricultural practices while also illustrating the bridging function of the concept.

TABLE 2 | The NBS framework for agricultural landscapes.

Essential function		Nature-based solution contributory mechanism	Indicative spatial scale of impact: Unit	Indicative spatial scale of impact: People	Indicative temporal scale of effectiveness
1. Sustainable practices *Must have a productive element*		1.1 Sustain or increase agricultural **production** by means other than standard approaches to the availability of water or nutrients, or plant breeding	Field	Household: income and food	Short to medium term
		1.2 Retain or increase available **nutrients** in soil, water and plants, in plant-or animal-available forms	Field	Household: income and food	Short to medium term
		1.3 Improve **microclimate** at the soil surface or in the cropping zone, by beneficial regulation of any combination of moisture, humidity, air movement or temperature	Field or landscape	Household: income and food Community: vulnerability and well-being	Short to long term
2. Green infrastructure *Must have a structural engineering function*		2.1 Regulate **water flows** (energy, rate or volume) on soil surfaces, in soil masses and at water body peripheries	Field or catchment	Household: income and food Community: vulnerability and health	Medium term
		2.2 Prevent **soil erosion** by armouring a slope or watercourse bank, or by catching eroding material (safeguard topsoil quantity)	Field or catchment	Household: income and food Community: vulnerability and health	Medium term
		2.3 Enhance **slope stability** against shallow mass failures by roots or other natural products increasing soil shear resistance, anchoring through failure planes and supporting soil masses by buttressing and arching (safeguard soil masses)	Field or landscape	Household: income and food Community: vulnerability and well-being	Medium term
3. Amelioration *Must have a beneficial biochemical, biological or microbial function*		3.1 Remove, degrade or contain **pollutants** in water, soil or air through any one or combination of natural physical, chemical or biological agents (bio- and phytoremediation)	Field or catchment	Household: income and food Community: vulnerability and health	Medium term
		3.2 Restore or stimulate beneficial **biota** for soil health, pollination or pest control, in the soil, cropping zone or nearby environment	Field	Household: income and food	Short to medium term
		3.3 Remove or store atmospheric **carbon** in soils or plants	Global	All societies: vulnerability to climate change	Medium to long term
4. Conservation *Must have a species preservation benefit*		4.1 Increase or protect **biological diversity** and habitat, either wild or modified (field scale)	Field	Household: income and food	Medium to long term
		4.2 Enhance **connectivity**, **area** or **health** of ecosystems (large scale)	Catchment or landscape	Community: vulnerability, health and well-being	Long term

Source: Landscape from ICIMOD (http://www.icimod.org/?q=rps_riverbasins). Wave (2.1) Abstract vector created by freepik - www.freepik.com. Landslide (2.3) original icon from www.clipartmax.com. Bee (3.2) and tree (4.1) original icons made by Smashicons from www.flaticon.com. Cow (3.3) icon made by Nhor Phai from www.flaticon.com. All other icons made by Freepik from www.flaticon.com.

A third discussion centered around a possibility of NBS being promoted regardless of the root cause of the problem. This, many argued, run the risk of making NBS technology-oriented and supply-driven, rather than outcome-oriented and demand-driven, and as such, legitimising "lack of capacity" as a barrier for adoption among reluctant farmers. In contrast, when farmers experience demand for a product, they often find their own ways to overcome technical capacity gaps. The motivation to accept uncertainties involved with changing practices can vary considerably, even among homogenous

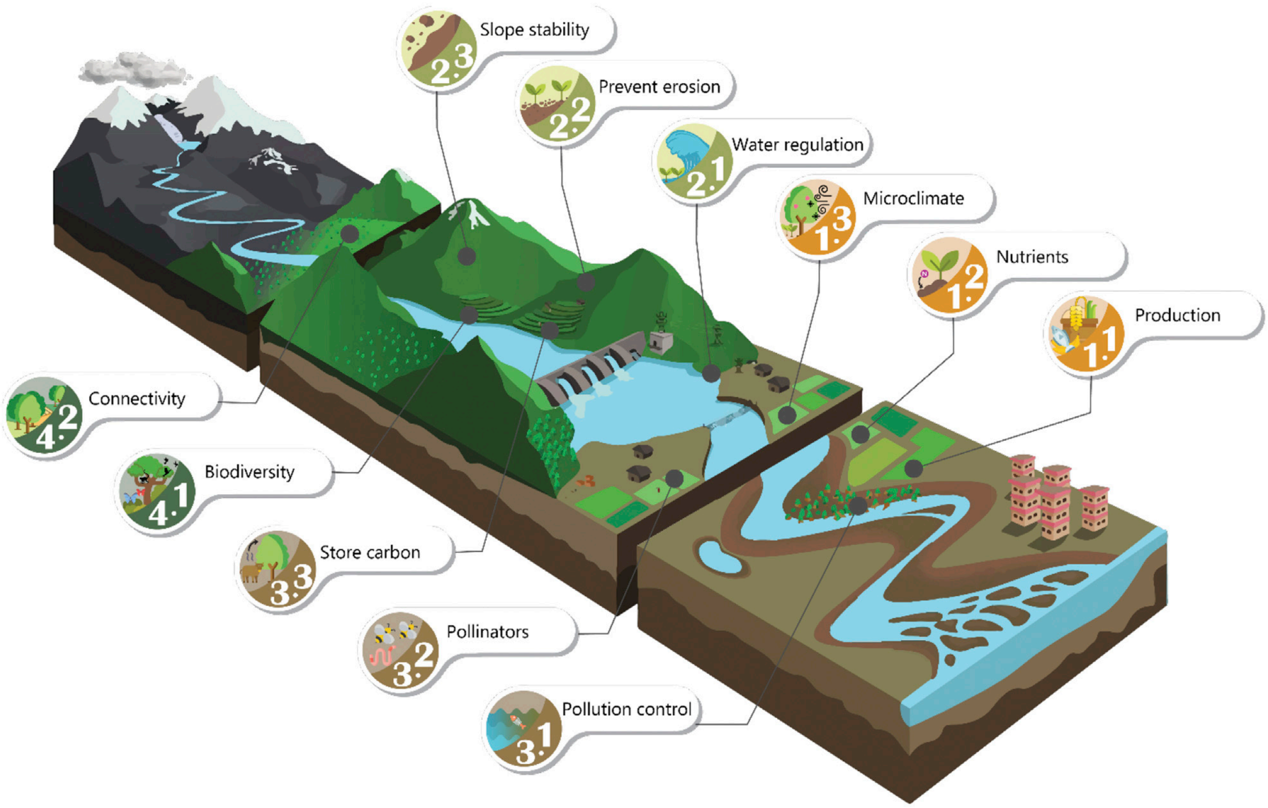


FIGURE 2 | NBS for agriculture landscapes. The corresponding numbers are found in **Table 2**. Source: Landscape from ICIMOD (http://www.icimod.org/?q=rps_riverbasins). Wave (2.1) Abstract vector created by freepik - www.freepik.com. Landslide (2.3) original icon from www.clipartmax.com. Bee (3.2) and tree (4.1) original icons made by Smashicons from www.flaticon.com. Cow (3.3) icon made by Nhor Phai from www.flaticon.com. All other icons made by Freepik from www.flaticon.com.

groups of farmers, as shown in a European case study (e.g. Gatto et al., 2019).

Other key points that were raised include whether NBS is considered as a disruptive solution, or whether it can work alongside industrialized monoculture to reduce negative ecological impacts; and the importance of establishing policy drivers and incentives for NBS including comprehensive economic costing to establish clear evidence of benefits to decision makers and establish commercial viability to the private sector and farmers. These issues are discussed in further depth in *Considerations for Implementation of NBS*. Each consultation concluded that a technical framework could be a helpful tool to identify and match nature-based “solutions” with immediate intentions against a longer-term view.

THE NATURE-BASED SOLUTIONS FRAMEWORK

This section outlines the framework with consultation feedback incorporated and provides explanation of the essential functions with examples of measures as emerged from the literature review. The full list of practices mapped to specific functions is stated in detail in **Supplementary Table 2**.

Framework Overview and Technical Dimensions

The framework (**Table 2**) builds on the use of NBS in response to challenges in agricultural landscapes (**Figure 2**), and consists of four essential functions that can be used to gradually add functionality, purpose and scale in project design (**Table 1**) with measures categorized according to their essential, or primary function:

1) Sustainable practices—primarily for production purposes, including natural nutrient and microclimate management, e.g. agroforestry and windshields. Anticipated benefits to people include more diverse and/or higher production quality, more stable productivity, safeguarded livelihoods, and reduced damage by temperature stress.
2) Green infrastructure—primarily for engineering purposes, including physical regulation of water and soil, and slope stabilization, e.g. grass strips, hedgerows, or terraces using natural material. Benefits include reduced damage by mass movement or additional fodder grass.
3) Amelioration—primarily for restoration of conditions for plants, water, soil or air and climate change mitigation, e.g. bio- and phytoremediation and mangroves. Benefits include

safe water, reduced health impacts stemming from biological pest control and carbon sequestration.

4) Conservation—primarily for maintenance or increase of ecological health at field or landscape scales, e.g. natural fallow or regeneration. Benefits include general well-being, cultural and spiritual benefits, safeguarded biodiversity, supported nutrient cycles, and increased resilience to environmental stress.

The production (1) and conservation-oriented (4) purposes frequently appear in the literature and easily lend themselves to be contrasted, such as in **Table 1**. "Conservation" in agricultural landscapes should be distinguished from "conservation" and protection of natural ecosystems. However, NBS are multifunctional and provide synergy benefits. For example, perennial systems could contribute to all four NBS-categories. Similarly, root causes to declining agriculture productivity can often be traced to neglected management in all four categories.

The establishment of an ecologically functional system can be achieved by systematically building up ecosystem functions through different components over time (succession) or joining areas (connectivity). As such, the framework can stimulate multidisciplinary action towards higher social and environmental outcomes. Spanning from the first category there is more focus on socioeconomic co-benefits, in companion with the second group many solutions can be implemented on small scale with return to the landowner/user. Further towards the third and fourth groups come increasing biodiversity co-benefits, the focus shifts more on planning for successions that 1) take longer time, 2) and/or require larger landscapes, and to build-up of natural buffers/conditions for those ecosystems to restore functions and return benefits to people. Such objectives are more effective through organised groups of land users/owners or communities, as socioeconomic benefits are possible, e.g. amelioration and carbon sequestration with multifunctional plants that generates non-(timber)-tree products. Similarly, there can be trade-offs involved between the four essential functions. For example, root causes to declining agriculture productivity can often be traced to neglected management in all four categories. If agriculture production is a subsidiary priority, or can be compensated for, the interventions can target larger spatial scales or aim for achieving higher environmental values, i.e. amelioration and conservation goals. The nature of some challenges may call for immediate action to avoid further environmental deterioration, such as removing toxic substances or reducing natural hazard risks.

The Functions

Sustainable Practices

First, "production"-oriented practices make use of the multiple ecosystem functions of trees, plants and (wild or domesticated) animals for agricultural production, while minimizing the negative environmental impacts of the production (Daryanto et al., 2018) such as regenerative agriculture and conservation agriculture.

For example, trees in alley cropping can play multiple roles: 1) tree crops for food and fodder production, 2) perennial alley crops, 3) trees for crop facilitation via shade, and 4) within-system tree diversity (Wolz and DeLucia, 2018). In agroforestry and sloping agriculture land technologies, in addition to production contributions, plants may also perform green infrastructure functions if, for example, planted as grass strips, or nitrogen-fixing legumes used as green mulch and fruit trees, planted along contours (McIvor et al., 2017; Are et al., 2018; Geussens et al., 2019).

Green Infrastructure

In the reviewed examples, GI practices were used for structural stabilization of slopes and controlling the flow of water and soil at field or catchment scale. Often GI entails the use of selected species which maximize their purpose such as root structure and morphology for erosion control, slope reinforcement or wave energy reduction. In the non-agriculture sphere, one main purpose of GI is disaster mitigation. Common examples were those of constructed wetlands for water regulation, storage and flood control. For example, in the United States and New Zealand, ecological infrastructure of wetlands with riparian forest, floodplains and constructed wetlands (Mander et al., 2005; Watson et al., 2016). Mangroves also have documented direct and indirect benefits for coastal protection and adaptation for both urban and rural livelihoods, small-scale fisheries, and ecosystems (Tran and Bui, 2013; Diop et al., 2018; Rahman and Mahmud, 2018).

Viewing NBS from the perspective of "design of new agroecosystems", we searched for evidence of engineered technologies essentially qualifying for multiple agriculture and non-agriculture purposes.

Agronomic Measures

When agriculture species plays the role of vegetation in GI, multiple functions are rendered. For example, grass strips control soil erosion and return crop yields (Are et al., 2018), where vetiver grass also can act as phytoremediation to trap phosphorous (Huang et al., 2019), or cut for animal feed. The efficiency of a catch crop also depends on physical elements, such as slope gradient (Novara et al., 2019) and root structure. Some papers related micro-terraces and built terraces as green infrastructure for agriculture (Zuazo et al., 2011; Liu et al., 2018). In northern India for example, simple weed strips and weed mulch also created micro terraces, which resulted in reduced soil erosion and higher yields (Lenka et al., 2017).

Engineering Structures

Agricultural waste can also be used as construction material for green infrastructure. For example, geotextiles made from local material such as bamboo, rice and wheat straw, and maize stalks were used to stabilize slopes in Lithuania, China, Thailand, and Vietnam, sometimes in combination with contour planting, with reported higher biomass production and crop yields, compared to no geotextiles (Bhattacharyya et al., 2012). There were no examples among the reviewed literature, but it is possible to imagine green slope stabilization measures on non-agriculture soils, providing non-timber forest products.

Amelioration—Phyto- and Bioremediation

Phytoremediation — the use of living green plants, and bioremediation — the use of microorganisms to break down or degrade contaminants, are considered cost-effective and environmentally friendly technologies for cleaning up polluted sites or preparing sludge before it is reintroduced to the environment. In the United States and Indonesia, a set of methods to control agricultural runoff, such as vegetated swales, enhanced stream buffers, denitrifying bioreactors, and constructed wetlands were referred to as GI (Anbumozhi et al., 2005), while according to our framework, their main functions place this measure into the category of amelioration. Many reviewed bio- and phytoremediation interventions were local, and studies therefore species-focused.

Bioremediation

The number of patents for new bioremediation technologies for water and soils are increasing at a fast rate, especially in China. A review showed that patents for using bioremediation agents, such as bacteria, enzymes, and fungi were more common than algae, plants and protozoa, as most patents targeted oil contaminants (Quintella et al., 2019). Specifically, in agricultural environments, anaerobic denitrifying bioreactors (hydraulic retention and biochar) can remove agricultural pollutants from farmland to surface waters, such as pesticides (Villaverde et al., 2018; Hassanpour et al., 2019). Of the 25 reviewed papers on bioremediation, most were concerned with removing nitrates, and with three Asian countries represented: China (5), India (1) and Pakistan (1). Within bioremediation, site selection and design are two important aspects. For example, denitrifying bioreactors require design that is resistant to differences in water flow during storm events to avoid leakage (Pluer et al., 2019). Among the literature featured many laboratory experiments, which suggests that this is an area where new and more advanced technology can be expected. Promising results were shown with rice straw instead of woodchip as carbon source in the bioreactor (Liang et al., 2015).

Phytoremediation

In phytoremediation, plants are purposely selected to extract pollutants from soil and water, or to exclude pollutants from biomass, or a combination of both (Jonsson and Haller, 2014). Through the search, we identified 14 studies on phytoremediation, most with the primary objectives being pollution control and desalinization. The extraction capacity of plants is important to inform about the potential use of remediation plants for feed or food. For example, to recover pesticide contaminated cotton soils in Nicaragua, scientists compared the distribution of persistent organic pollutants in different vegetative organs in three cultivars of amaranth. Overall, although the type and amount of pollutant that each cultivar extracted from the soil varied significantly, parts that could provide feed, stems and leaves, accumulated higher concentrations than the roots and seeds (Haller et al., 2017). The uptake and translocation of antibiotics in maize is another example of the potential use of agricultural crops for phytoremediation (Zhang et al., 2019). A recent review illustrates the efficiency of agriculture and forestry plants in metal extraction from mercury-contaminated soils and water, and also risks of accumulation in edible tissues for animal and human health (Tiodar et al., 2021).

In constructed wetlands, different riparian vegetation types such as coniferous, deciduous broad leaf or evergreen broad leaf forests, aquatic or herbaceous plants play different roles that are designed for controlling and managing water pollution (Wang et al., 2018). For some purposes, phytoremediation in constructed wetlands may perform better together with other technologies for removing toxic agrochemicals, such as bio-mixtures for biopurification (Gikas et al., 2018). Functions of riparian zones and buffer strips and their designs are described by Mander et al. (2017). Specifically, the width of the vegetated buffers, which may vary between 1 and 4,000 m, matters for protecting water sources and crops against pesticides depending on the habitat — something which is not reflected in legal documents (Gene et al., 2019).

Climate Change Mitigation

While many practices have production or conservation purposes (e.g. FAO, 2016a; Zomer et al., 2016; Hernández-Morcillo et al., 2018; Rosenstock et al., 2018), their contributions to climate change mitigation appear underestimated in the reviewed NBS and GI practices—or conversely, were seldom referred to as NBS or GI in the literature search. However, when tree planting for carbon sequestration comes at the cost of biodiversity and local rights to resources, it is a distraction from the meaning and intentions of NBS (Seddon et al., 2021). One exception was hedgerows, which increase soil organic carbon but often struggle to get recognition as a mitigation contributor (Hernández-Morcillo et al., 2018).

Possible explanations, despite numerous policy and funding mechanisms, could be that the scale of interventions necessary for a significant global impact is difficult to monitor, conflict with landscape diversification, or compete with other land uses and ecosystem goods and services (Namirembe et al., 2015; Cohen-Shacham et al., 2016).

Conservation

For the conservation category, in landscapes with human impact the main purpose is to build up connected ecosystems, ecosystems functions and biodiversity, temporarily such as natural fallows, long-term or permanently, such as natural forest regeneration. Various landscape approaches aim to achieve multiple goals from *ecological intensification* of crop production with biodiversity focus (Garibaldi et al., 2019) to ecosystem services within payments for ecosystem services (PES) schemes (Holt et al., 2016; Karabulut et al., 2019). One particular intention with practices under this essential function, is to ecologically connect conservation agriculture on field-units across larger landscape mosaics in landscape approaches (Holt et al., 2016).

The review illustrated that integration of practices can connect patches in the landscape. First, in Europe with functional agrobiodiversity approaches, where permanent grassland and crop diversification within ecological focus areas involved a certain percent of arable land that was set aside to be used for field margins, hedges, trees, fallow land, landscape features, biotopes, buffer strips, and afforested area (Delbaere et al., 2014). Similarly, connectivity was achieved with ecological infrastructures, such as woodland hedges, rosaceous hedges, grass strips, wildflower strips, and field margin (Rosas-Ramos et al., 2018). In Pakistan, an example of EbA included connecting landscapes through practices such as crop rotation, intercropping, agroforestry, crop diversification, live fencing, and wind barriers by trees (Shah et al., 2019). These examples show that many biodiversity conservation practices also contribute to ameliorative functions, such as carbon sequestration and pollinators (IPCC, 2019), that build up multiple ecosystem values over time.

Temporal and Spatial Scales: Sequencing, Successions, and Connections

Foreseen and unforeseen risks affect land use decisions across spatial and temporal scales. As conservation challenges are rarely foreseen, "best practice" solutions, which denotes predictability, are ill-suited for complex systems (Game et al., 2014). Instead, NBS need to be designed as a series of interventions to reinforce the resilience of ecosystems in order to prevent, reduce, respond to, or adapt to existing or anticipated stressors. An important aspect of the "conservation" function in the framework is therefore the process of connecting or expanding NBS-measures to cover larger timescales and areas of the landscape. These scales can be considered as a mosaic onto which we may overlay physical disruptors, e.g. environmental degradation, invasive species, pest and disease pathways, and interventions that connect landscapes, e.g. biodiversity corridors and constructed wetlands.

Prioritisation includes identifying the sequencing order for a stable succession. For example, natural regrowth and root development in riparian wetlands take years (Frątczak et al., 2019) and the full effect of trees for slope stabilisation comes decades later (Stokes et al., 2010). Timing the interventions thus depends on natural regeneration processes, as well as when implementers expect to see certain benefits. The benefits (or dis-benefits), and urgency of them, can be perceived and prioritized differently by certain groups at various scales from field-farm-farmer to landscape-ecosystem-community scales (UNEP, 2021). Important aspects of successful NBS involves responsive decentralized management (Game et al., 2014), removing barriers that focus on short-term economic returns to cover investments and to focus on an affordable succession of NBS practices that pay back over time.

CONSIDERATIONS FOR IMPLEMENTATION OF NATURE-BASED SOLUTIONS

This section moves from consolidation of definitions of practices under the framework, to considerations for implementation. Factors and gaps emerging from the literature are discussed, as well as insight which emerged from the consultation workshops.

Economic Dimensions

The economic argument for adoption, which was stressed in consultations from the dual perspectives of farmers and decision makers of developing countries was not reflected in the literature. Ten reviewed papers included economic assessments of the practice itself or of the environmental values of the practices. Among these are economic estimates calculated on management approaches to reduce sediment loads (Mtibaa et al., 2018) and agriculture runoff (Gikas et al., 2018; Irwin et al., 2018). A study in Tunisia by Mtibaa et al. (2018) found that while contour ridges alone halved the sediment yield, the most cost-effective option was a combination of practices, including buffer strips, conversion to orchard, and grass strip cropping. Similarly, Gikas et al. (2018) showed that two low-cost options with plants in constructed wetlands, performed better when combined with bio-mixtures containing coconut fibre for biopurification. Other estimates, such as those by Irwin et al. (2018), related the improvement in water quality from reduced agriculture runoff with an associated value for residents and recreation users. Here, ten percent improved water quality resulted in a "lifetime cost benefit ratio" of 2.9.

Shortcomings in economic assessments can be attributed several issues:

First, difficulties in correctly evaluating ecosystem values. For example, the effects and valuation of agroforestry ecosystem services were clearer at the farm/plot scale, whereas attribution easily got blurred in the mixed land uses at landscape scale (Kay et al., 2019). The scales add challenges when negotiating economic and socio-cultural stakes in landscapes with diverse tenure and management.

Second, difficulties extrapolating results from smaller empirical studies, e.g. the role of pollinator services for global scale food production. To overcome this, Melathopoulos et al. (2015) devised an approach to estimate values of pollinator services from three different assumptions: 1) the degree of dependency of crops on pollinators; 2) pollinators need different habitats and pollinate different crops (wild versus domesticated) hence the cost to retain them will vary; 3) whether the price of the ecosystem service is aligned with the risk, e.g. the value depends on the probability of a bee pollinator collapse.

Third, underlying economic assumptions of grey versus green infrastructure depend on how risk, investment costs and value of losses are calculated. For example, Onuma and Tsuge (2018) tried to determine when green infrastructure is preferable to grey for disaster risk reduction. This was done by developing parameters to compare the two options in view of hazard, population potentially affected, and associated vulnerability. Although their primary focus was not on agriculture, similar valorization principles can have applications for GI in agriculture. For example, grey infrastructure is designed as a defense to one particular natural hazard and breaks at a certain magnitude, while mixing grey with green infrastructure can be more durable.

Additionally, costs are often lower for recovering green infrastructure after a disaster event.

Lastly, NBS interventions need to consider surrounding land-use change, such as increasing rents on intensive agriculture land, which will likely drive costs for conservation and carbon credit compensations (Phelps et al., 2013). One review pointed out that many studies, especially in developing countries, fail to specify baseline conditions to which cost-effectiveness evaluations are made. This is partly due to a shortage of available georeferenced data on agriculture management, costs and prices (Ovando and Brouwer, 2019). Data shortage also risks misinterpreting conservation *vis-a-vis* production interests (sparing versus sharing debates), where the historical management contexts are required to understand the ecological values and trade-offs (Angelstam and Lazdinis, 2017; Naumov et al., 2018), not the least in the light of potential tenure issues (FAO, 2016b; Carter et al., 2017; Borelli et al., 2019). Furthermore, the ongoing rapid land-use changes across Asia (Tenneson et al., 2021) may make it difficult to determine a baseline or an "ecological equilibrium" to reflect "ecological health." More studies involving a long-term lens on economic assessments can contribute to better estimates of avoided loss and damage by NBS and similar interventions and stimulate adoption.

Social Dimensions and Long-Term Adoption

Several studies in the NBS review indicate that farmers may not adopt sustainable practices despite having witnessed ecosystem benefits, because of increased initial costs, labour inputs, or customs and preferences (Chapman and Darby, 2016; McWilliam and Balzarova, 2017; Cerdà et al., 2018). To overcome this, farmers' willingness to adopt new practices can be influenced by presenting cost-benefit assessments of different management options. Examples included cover crops in various ecosystems (Daryanto et al., 2018) and a system-dynamics modelling study on paddy field management from Vietnam, where the dynamics between farmers and their rice agriculture operations were integrated with the role of fluvial sediment deposition within their dyke compartment (Chapman and Darby, 2016). The latter study found that triple-cropping was only optimal for the wealthier farmers and in the short-term, while sluice gate management to enable soil nutrient replenishment would be a more economically and environmentally sustainable practice.

Despite a vast body of literature concerned with piloting different types of compensations for land use conversion, particularly PES, few mentioned NBS. In Uganda, Geussens et al. (2019) investigated farmers willingness to accept eight practices (qualifying as sustainable production or GI in this framework: i.e. minimum tillage, mulching, contouring, trenches, grass strips, agroforestry, and riverbank protection) under nine different compensation levels, or PES contracts. The study drew two important lessons for NBS. First, the biggest difference between willing and reluctant PES-adopters, concerned their perceived benefits. Their preferences depended on the intervention, the compensation level, and whether they received community funds or individual compensation. Second, project designers contrasted willingness to adopt and the reduced effectiveness of scattered practices. Hence, a minimum number of farmers were required for landscape benefits. The willingness to accept was high when the need for a different solution had reached a certain threshold, such as severity of degradation (the Uganda example), or when farmers have run out of other viable options. Ultimately, PES schemes would benefit land uses with high ecosystem values by combining marketable and non-marketable ecosystem services, such as biomass production and groundwater, soil quality, carbon sequestration, or penalizing land-uses with dis-benefits (Kay et al., 2019).

Illustrating complex trade-offs in transparent ways can help to reach negotiation solutions. For instance, Rosa-Schleich et al. (2019) reviewed the economic and environmental trade-offs among nine diversified farming practices. For each practice, they developed a matrix of ecological and economic benefits, which were converted into two axes. The space showed what clusters of practices were perceived to give high ecological benefits (agroforestry), high economic benefits (structural elements), or high in both (organic agriculture). Similarly, for the purpose of restoring an environmentally degraded mangrove ecosystem in Bangladesh, scientists developed a relative environmental and economic matrix with a quantitative cost-benefit study on four silvo-fishery systems under different restoration scenarios: integrated mangrove-shrimp, crab-mangrove, mangrove bio-filtering, and nypa-shrimp over three periods between 0 and 10+ years (Rahman and Mahmud, 2018). Both studies showed that combinations of practices with multiple functions were beneficial, particularly when the introduction of structural elements have insignificant economic or productive motives. Moreover, interventions that require decades to mature, such as mangrove restoration, also strongly depend on community participation and governance commitment (Rahman and Mahmud, 2018).

Policy Dimensions

Among the evidence for long-term adoption and transformation, the review raised examples of where NBS-practices were embedded in institutional and policy decisions that went beyond subsidies and conservation goals. For example, Albert et al. (2017) identified four premises for economic valuation of ecosystem services: 1) an institutional analysis to establish uses of nature and incentives of different stakeholders, 2) cost-and-benefits associated with the change in nature, 3) public and private sources of incentives to land managers, and 4) trade-off assessments between societal goals to establish winners and losers coming with the policy package, in their case the Common Agriculture Policy. The benefit of long-duration policies was shown similarly in an 18 year-long study from Italy, which concluded that through a persistent government policy, the different needs of different farmer typologies could be met, from early to late adopters (Gatto et al., 2019). Their study on implementing and maintaining hedgerows, reported that early adopters required that the compensation could be integrated with their income-generating activities, while the next group of adopters were those who received support to plant new hedgerows rather than those who maintained their existing ones. The third phase of adopters were motivated by social pressure and public acknowledgement of farmers' work, and the late adopters followed when they felt pressure from neighbor farmers rather than the public. The role of governments for setting policies and long-term pathways is repeated also for regulating public goods where PES-markets are limited, such as fish and fish habitats (Mulazzani et al., 2019).

Some studies found that blanket policies fail to reflect the complex realities and trade-offs (Holt et al., 2016). The consultation workshops generated more practical insights to this literature. First, underlying causes of farmers' reluctance, such as control over resources, are rarely addressed and instead generally "solved" by training and sensitization. For example, tenure insecurity is known to restrict smallholder farmers' longer-term investments in diverse perennial farming systems (Borelli et al., 2019). Second, existing governance barriers, such as rigid policies and institutional silos, were overlooked in many studies. Such barriers can demotivate both decisionmakers and grassroot initiatives. For example, a structured analysis within seven Indonesian government institutions identified broad gaps and inconsistencies for institutionalizing valuation into policy (Phelps et al., 2017). Third, the workshop participants were largely in agreement that sufficient, stable and long-term support was lacking at the landscape-scale NBS across Asia. Exemplifying the importance of this as a precondition included Vietnam's national PES policy, which after almost a decade of implementation still has difficulties reaching impact at scale. Among the reasons raised were that no compliance is required, and the net benefits are so low (fixed, non-negotiable compensation) that often only community-based payments are viable to payout. Moreover, while community compensation is often preferred by the poorer households, this is unlikely to motivate adopters in the long run if living standards improve. Incentives and policies to change from short to long-term sustainable behaviors are urgently needed, notably from government or companies buying the products. The Uganda case suggested, that since PES compensations are generally low and may be subject to changing compensation levels, (wealthier) farmers who do not need payments, should not receive them even if they make interventions (Geussens et al., 2019). Decision support tools seem to be used in the initial stages of research projects, while the review gave little evidence for them becoming permanently integrated in decision processes. Four papers concerned tools for negotiating human-environmental-governance relationships, typically trade-off models for anticipating or assessing policy impacts on ecosystems (Rega et al., 2018; Karabulut et al., 2019).

Nature-Based Solutions as a Disruptive Solution

The consultation workshops confirmed an urgent need for system-level interventions in agriculture that can effectively address multiple challenges simultaneously. One concern expressed in the first consultations was that some (decision makers, private sector interests) may view NBS as a troublemaker if promoted as a replacement for industrialized monoculture or "grey" infrastructure. Some argued, if NBS can appear alongside monocultures, it could gradually and more easily be "mainstreamed" into large-scale agriculture landscapes, such as rice-cultivated deltas, to mitigate some of the most harmful impacts. Others commented that such entry points would limit opportunities to fully use nature to restore ecosystem services, such as providing habitat for pollinators and natural predators. Following the need to address complex challenges, the consultations indicated that one selling point of the NBS framework is to demonstrate how to break spirals where agriculture cause environmental problems (e.g. overuse of agrochemicals spilling into waters) which create new problems for agriculture production (e.g. polluted soils and water impacting on pollinators and food safety), and how these problems are connected across landscapes. Few of the reviewed papers made substantial references to how NBS interventions could contribute to international commitments. Conceptually, the NBS-framework provides entry points to harmonize goals of several UN Conventions, such as on climate change, land degradation, biological diversity, and Sustainable Development Goals.

A concrete example to bypass two persistent obstacles for adoption: financial support and technical knowledge, could be to use decision support tools for comparing when GI is preferable to grey infrastructure (Onuma and Tsuge, 2018). This can be translated into loss and damage recommendations from, for example, economic assessments of benefits from GI for flood control (Watson et al., 2016), or post-disaster assessments of impacts on watershed services and water security (HLPE, 2019). Like the consultations pointed out, higher level public officers may be motivated to co-invest in implementation if NBS can attract private investments (FOLU, 2019). More importantly, to sustain long-term effects of NBS and GI, studies often highlight governance and the role of community, private and public sector engagement (IUCN, 2020; Monteiro et al., 2020; Dumitru et al., 2021). The nature of such relationships is fundamentally diverse across the globe, and each setting need to find their own new modalities.

Transboundary Challenges and Opportunities of Nature-Based Solutions

The literature review did not present solutions to the transboundary nature of many challenges, especially water-related ones, although many NBS examples seem fit for such purposes. Certain lessons can be drawn from catchment projects, such as PES, about acceptable compensation levels and their duration. Furthermore, successful NBS implementation will likely benefit from breaking up some institutional silos. This requires a common terminology and international policy frameworks. To illustrate this process is the development of ASEAN agroforestry guidelines, where ministers agreed on a regional strategy. Subsequent work nationally is described in Catacutan et al. (2018) and Singh et al. (2016). Further, NBS overlap with some of the Committee on World Food Security's principles on guiding frameworks on rights, livelihoods and tenure (CFS, 2014). For instance, in relation to Principle 6 "Conserve and sustainably manage natural resources, increase resilience, and reduce disaster risks," NBS can represent a set

TABLE 3 | Categories of needed actions and possible concrete examples.

Action	Concrete examples
Develop diagnostic assessment tools with applied assessments of key landscapes, to identify where there is potential to implement NBS. Tools need to be flexible enough to capture the contexts for NBS over space and time, including trade-off analyses of winners and losers, impacts on agriculture production and on natural ecosystems.	The NBS framework presented here, and an NBS Planning Tool (to be developed in separate publication), are provided as initial tools that can be further adapted. Development of practical guidance for implementation of NBS, based on diagnostic assessments.
Identify and agree upon landscapes to target for NBS applications particularly landscapes with high levels or risk of agroecosystem degradation based on agreed intervention criteria and potential for NBS adoption.	Review the status of degradation across agricultural landscapes and prioritize sectors with the highest environmental costs for NBS interventions. Apply NBS diagnostic assessments in the preparation of project design exercises targeting restoration of agroecosystems
Set up multidisciplinary networks with ongoing NBS sites for application and demonstration of the NBS framework and related approached. and including awareness raising activities, capacity building and exchange tours.	Use participatory integrated landscape designs and simulations to help to build up functional ecosystems with values that also motivate land users over time. Create dialogue platforms for value chain actors to understand how NBS approaches can deliver wider value for value chain level recognition (e.g. branding or product narratives) and resilience
Implement complementary NBS approaches via action research, participatory experiments and scaled-up actions to complement existing development projects and loans with an NBS outlook.	Participatory, multidisciplinary integrated landscape designs and simulations to help to build up functional ecosystems with values that also motivate land users over time. Integrate indigenous knowledge and approaches into a suite of NBS options for agriculture.
Establish regular longitudinal monitoring and reporting systems for NBS-sites to study on-site and peripheral impacts, (before) during and after project completion, including reporting on people's indicators of wellbeing. Monitor benefits and disadvantages of larger adoption of NBS over different spatial and temporal scales.	Set up phytoremediation recommendations to prevent agriculture runoff into waters and reservoirs, for different problems and with species for different purposes, e.g. compost, feed, bioconstruction material. Measure the change in labor inputs. The NBS Monitoring Tool is an initial tool that can be further adapted.
Where relevant, link NBS work in agriculture to policy processes including national policy priorities linked to the SDGs as wells as global processes on NBS such as IUCN's NBS standards and the NBS Initiative	Develop cost/benefit analysis of NBS applications in agriculture to allow for easy comparison of NBS and traditional approaches Organize policy consultations to identify and review purposeful qualification criteria and indicators of NBS for agroecosystems. Ensure local indicators contribute to national reporting targets, e.g. NDC.
Identify ways to scale-up NBS via traditional, public funds and innovative financing mechanisms.	Set up competitive start-up or innovation funds for your agri-entrepreneurs to invest in new marketable nature-based solutions.

of environmentally sound practices that also can reduce the negative impacts of agriculture. Additionally, a stronger rights and co-investment perspective can be added to the NBS framework from the Responsible Investment in Agriculture and Food Systems (RIAFS), which offer a set of non-binding principles to promote responsible investments that specifically contribute to food security and nutrition.

Proof of Evidence From the Top Down and Bottom Up

The consultations pointed out that proof of evidence was viewed vital for the initial adoption of NBS. Details of such evidence must be worked out with various stakeholders in an agricultural landscape (**Table 3**), as the interests and motivation vary among land owners and users, decision makers, private sector, and the public.

Approaches need to accommodate both stable policies that motivate change *and* community engagement that ensures local problems are addressed. Although positive spill-over effects on adoption were noted over time in some European studies (e.g. Gatto et al., 2019), prerequisites for NBS-adoption outside European contexts need to be better understood. Collecting good practices could aim to fill specific data gaps on e.g. measurable benefits and ecological health. As evidence is generated, what counts as NBS will likely continue to evolve over time. Allowing a credible degree of flexibility within a concept is necessary, as an over-reliance on best practices recommendations can hinder creativity, co-learning, and may result in maladaptation (Game et al., 2014; Schipper, 2020; Eriksen et al., 2021). Furthermore, it was discussed whether landscape diversity requires a certain degree of homogeneity or heterogeneity among farmers, farm sizes or their activities. To opt for scaling of best practices may not always be desirable or achievable given the diversity of situations and problems in any specific agricultural area and community. Therefore, there was strong agreement among consultation participants that NBS need specific entry points to pursue opportunities to transition from short to long term impacts. For instance, through environmental economics accounting, "green GDP," or capping a maximum for environmental debts that can be moved into the future. Another entry point was "urgency triggers," as certain practices may only be adopted once a certain ecological (or economic) state worsens in a location or group, such as after a disaster, when human and environmental health needs demanded or were pushed by consumers or farmer organisations. Urgent entry points relate to the importance of a well-established baseline and setting common goals and success indicators—all essential parts of planning tools. One suggestion was that NBS-landscapes can be planned where a minimum level of "success" of NBS can be considered when resiliently building up vital ecosystem functions while delivering the social and ecosystem benefits people expected.

More transparent value chains were seen a precondition, where social media was perceived a tool to remove some distorted market information, especially when the policy development process was too slow. Equally important to identifying entry points, is the development of a common vision for NBS as part of broader efforts to support more sustainable and resilient food systems. This framework for nature-based solutions in agricultural systems is a response to a gap in available tools and guidance on how NBS can be applied to the agriculture sector.

This NBS framework is designed to provide policy makers and practitioners with guidance to develop inclusive, multi-purpose and nature-positive solutions to support the improved management and long-term sustainability of agricultural production systems. It is currently being tested with a companion project planning tool by FAO in five South and Southeast Asian countries to facilitate such intentions.

Looking ahead, **Table 3** outlines categories of needed actions and possible concrete examples based on consultations. The framework developed in this report can provide needed guidance to inform this work.

CONCLUSION

The literature review and case studies presented at the regional workshops indicated that NBS approaches to date have been small in scale and focused on marginal lands at the fringes of major production landscapes. Empirical evidence on NBS and GI for agroecosystems is biased to western contexts. Few reviewed papers presented evidence of socioeconomic benefits of NBS.

The consultations identified limits to and potentials for adoption of the framework in major production landscapes with significant agroecosystem degradation. The consultations recommended that planning of successions is critical for achieving resilient impacts at scale and over time, to 1) select and sequence what and how to intervene to generate positive biophysical interactions and social benefits in and between agroecosystems, and 2) sustainably expand connectivity of positive interactions. In developing contexts, a gradual approach, based on decentralized piloting and demonstration of NBS approaches in a range of ecozones and socioecological contexts, would allow a mosaic of small-scale cases to be connected through a process of exchange and adaptive learning via networks and ecological interconnectedness. Such gradual approach would build up much needed evidence from practices and landscapes on the scalability of best practices and how to adapt NBS principles for implementation in developing countries.

To be effective, NBS in agriculture will require the identification of entry points with the support of a wide range of actors in the production landscape (farmers, communities and resource managers, local government extension workers and advisors at farm and landscapes scales, downstream value chain actors at local and global levels and national policy makers). Partnerships of actors, public and private, based on mutual interest in restoring major production landscapes through NBS are needed to ensure a wide support and the most potential to lead to lasting change in management practice. Policies can support the long-term commitments needed for restorative NBS approaches. The NBS-framework can facilitate the documentation of promising designs and practices for an overarching program of action. The next steps in testing the application of this framework involves reviewing evidence from Asia on the potential contributions of NBS to national polices for climate resilient agriculture, land restoration, biodiversity and sustainable development targets.

AUTHOR CONTRIBUTIONS

JC-R and BD conceptualised the original idea. BD, JC-R, MC, JH, and ES led consultation workshops. ES led the research and lead-authored this manuscript. JH, CP-G, HT, and MV conducted the review. CP-G, JH, and MV prepared the figure and tables. All authors contributed to analyse results, develop the framework, and write the paper.

FUNDING

This research was funded by the Food and Agriculture Organization of the United Nations (FAO) under a regional programme on “Identifying nature-based solutions (NBS) and green infrastructure (GI) in the agriculture sector for more resilient rural communities in Asia”.

ACKNOWLEDGMENTS

The authors are sincerely grateful for the critical and constructive comments from consultation participants on the technical report leading to this article. The authors also wish to acknowledge two reviewers for valuable comments which have improved the manuscript.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2021.678367/full#supplementary-material

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Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Environmental Science

REVIEW
published: 14 October 2021
doi: 10.3389/fenvs.2021.723166

Nature-Based Solutions for Urban Climate Change Adaptation and Wellbeing: Evidence and Opportunities From Kiribati, Samoa, and Vanuatu

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OPEN ACCESS

Edited by:
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Reviewed by:
Carlo Giupponi, Ca' Foscari University of Venice, Italy
Nathaniel K. Newlands, Agriculture and Agri-Food Canada (AAFC), Canada

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Specialty section:
This article was submitted to Interdisciplinary Climate Studies, a section of the journal Frontiers in Environmental Science

Received: *10 June 2021*
Accepted: *17 September 2021*
Published: *14 October 2021*

Citation:
Kiddle GL, Bakineti T, Latai-Niusulu A, Missack W, Pedersen Zari M, Kiddle R, Chanse V, Blaschke P and Loubser D (2021) Nature-Based Solutions for Urban Climate Change Adaptation and Wellbeing: Evidence and Opportunities From Kiribati, Samoa, and Vanuatu. Front. Environ. Sci. 9:723166. doi: 10.3389/fenvs.2021.723166

Climate change and urbanisation in combination put great pressure on terrestrial and ocean ecosystems, vital for subsistence and wellbeing in both rural and urban areas of Pacific islands. Adaptation is urgently required. Nature-based solutions (NbS) offer great potential, with the region increasingly implementing NbS and linked approaches like ecosystem-based adaptation in response. This paper utilises three Pacific island nation case-studies, Kiribati, Samoa and Vanuatu, to review current NbS approaches to adapt and mitigate the converging resilience challenges of climate change and urbanisation. We look at associated government policies, current NbS experience, and offer insights into opportunities for future work with focus on urban areas. These three Pacific island case-studies showcase their rich cultural and biological diversity and, importantly, the role of traditional ecological knowledge in shaping localised, place-based, NbS for climate change adaptation and enhanced wellbeing. But gaps in knowledge, policy, and practice remain. There is great potential for a nature-based urban design agenda positioned within an urban ecosystems framework linked closely to Indigenous understandings of wellbeing.

Keywords: nature-based solutions, climate change adaptation, wellbeing, traditional ecological knowledge, urbanisation, Kiribati, Samoa, Vanuatu

INTRODUCTION

Nature-based solutions (NbS) are defined by the International Union for Conservation of Nature (IUCN) as "actions to protect, sustainably manage, and restore natural or modified ecosystems, that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits" (Cohen-Shacham et al., 2016). More simply, NbS are solutions to societal challenges that involve working with nature. They aim to enhance the resilience of ecosystems, their capacity for renewal and the provision of ecosystem services

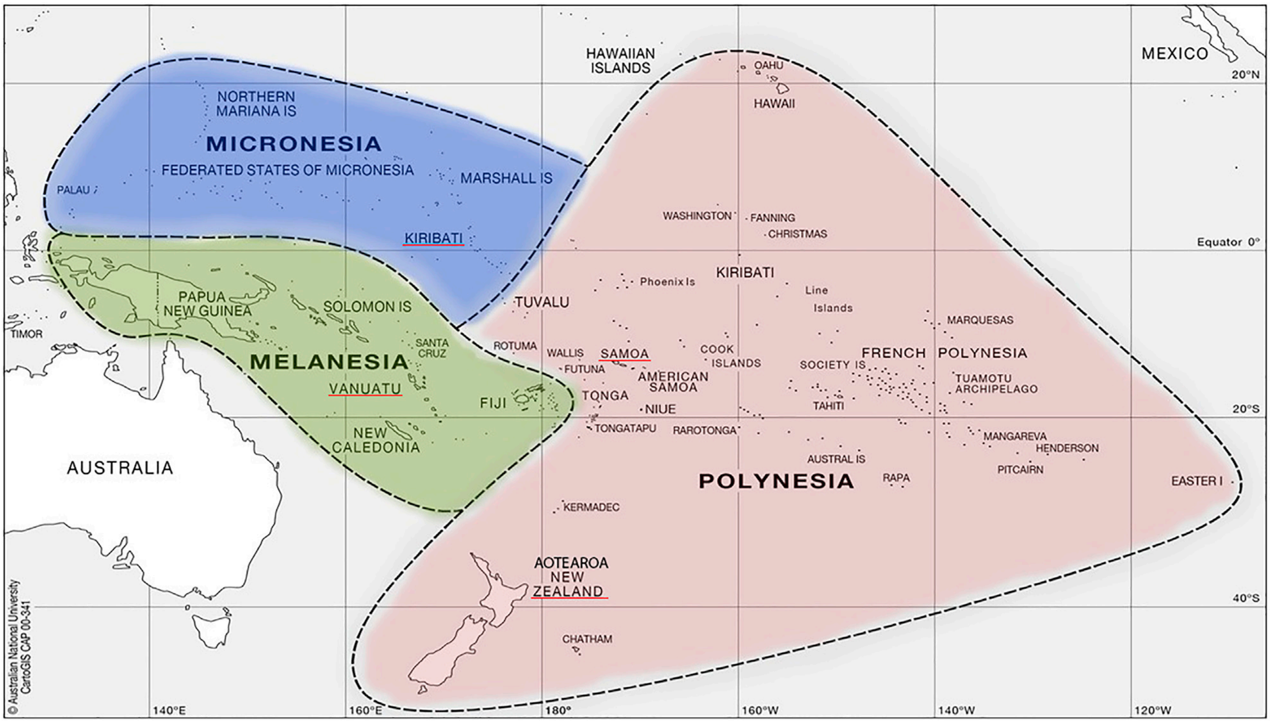

FIGURE 1 | Regions of Oceania. Source: Adapted from Australian National University (CC Licence).

(International Union for Conservation of Nature (IUCN), 2021). NbS have gained popularity globally as an integrated approach for responding to climate change, biodiversity loss, and broad sustainable development challenges. To date, for example, more than 130 countries have included NbS actions in their national plans under the Paris Agreement (International Union for Conservation of Nature (IUCN), 2020a). Critically, however, substantial gaps in the NbS practice evidence base remain, with much more work focused on Europe and other parts of the Global North, despite nations and communities in the Global South, including small island developing states, being more vulnerable to climate impacts (Chausson et al., 2020). In addition, there is also a strong need to increase the evidence base for NbS in urban areas (Kabisch et al., 2016).

The Pacific islands region (see **Figure 1**) is confronted by the twin "mega-trends" of climate change and urbanisation (Trundle et al., 2019). The impacts of climate change and associated increased and intensified extreme weather events in the Pacific region are well documented. In addition, the urbanisation rate across Pacific island small states has increased from 22.5% in 1960 to 39% in 2019 (World Bank, 2021), with further increases inevitable. Overall, climate change and urbanisation combine to place increasing pressure on interconnected island-based terrestrial and ocean ecosystems vital for subsistence, livelihoods, and wellbeing. Accordingly, Pacific island governments are increasingly prioritising NbS and particularly linked approaches such as ecosystem-based adaptation (EbA) in their national climate change policies and associated government priorities. This work, at different geographic scales, is supported by a range of development partners such as IUCN, the Secretariat of the Pacific Regional Environment Programme (SPREP), the Pacific Community (SPC), the United Nations Economic and Social Commission for Asia and the Pacific (UNESCAP), international and local non-governmental organisations, bilateral donors such as the aid programme of the New Zealand Ministry of Foreign Affairs and Trade, and other multilateral organisations.

We utilise three national Pacific island case-studies; Kiribati, Samoa and Vanuatu, to review current NbS and EbA approaches to adapt and mitigate the converging resilience challenges of climate change and urbanisation. We firstly introduce NbS and EbA and the potential benefits they offer, including as strategies for adaptation in urban areas. We then focus on the island case-studies; introducing the context and then reviewing current NbS and EbA case experience, linked government policy, and discussing implementation challenges and opportunities. In doing so, we highlight the importance of traditional ecological knowledge (TEK) driving NbS and EbA approaches so that they are appropriate and effective for Pacific islands. We also introduce ongoing research, focused on developing a nature-based urban design agenda for Oceania (including Aotearoa New Zealand). This work is positioned within an urban ecosystems framework closely linked to TEK and Indigenous understandings of wellbeing. This is vitally important if NbS are to be grounded locally and thus more likely to be effective both ecologically and culturally.

NATURE-BASED SOLUTIONS IN URBAN AREAS

NbS aim to produce multiple societal, cultural, health and economic co-benefits for people while conserving or generating increased ecological health. Inherent in NbS is the acknowledgement that the health of ecosystems and the biodiversity contained within them is essential for human survival. NbS acknowledge that working with nature, rather than against it or without it, can lead to more effective, economical and culturally appropriate solutions to societal challenges while concurrently conserving or restoring biodiversity (Pedersen Zari et al., 2019). NbS also bring, or offer potential for, multiple other benefits. UNESCAP highlight, for example, that NbS: 1) provide cost-effective environmental, social and economic benefits; 2) can support communities, both rural and urban, in accessing natural resources and using them sustainably to support livelihoods; 3) can build from traditional ecological knowledges; 4) and revitalise cultural connections to nature to raise awareness, educate, and engage urban communities (UNESCAP, 2019).

NbS is an umbrella term for several other concepts growing in use in related professional communities, academic discourse and policy debates such as: EbA; natural climate solutions, ecological restoration; ecological engineering; urban green and blue infrastructure; ecosystem-based mitigation; ecosystem-based disaster risk reduction; natural capital; forest landscape restoration; and potentially biomimicry and biophilic design (Griscom et al., 2017; Raymond et al., 2017). Overall, the precepts fundamental in unifying the NbS concept are: 1) an understanding of the benefits that humans derive from ecosystems and the services that they provide; 2) an acknowledgement that people can learn from nature; and 3) recognition of the strategic importance of strengthening ecosystem health and human relationships with ecosystems to increase human wellbeing and society's ability to adapt to various changes. A wide range of activities can be categorised as NbS. In Oceania, for example, UNESCAP profile the rehabilitation of mangroves (for coastal protection and also biodiversity benefits), combining natural and engineered infrastructure for water management, urban agroforestry and gardening, the establishment of Educational Managed Marine Areas, and rehabilitation of wetlands and forest landscapes (UNESCAP, 2018). However, this is only a limited list. Many other activities can be categorised as NbS, broadly encompassing greenhouse gas reduction, flood and erosion control, coastal defence, cooling/shading, food and water security, water quality improvement, vegetation and habitat restoration, and the integration of built infrastructure including buildings with ecosystems, particularly in urban settings. Seddon et al. (2021) note that to qualify as NbS an action must provide one or more benefits to humans while causing no loss of biodiversity or ecological integrity compared to the pre-intervention state. Ideally, there should be ecosystem improvement–hence a generally strong focus on ecosystem restoration inherent in NbS.

It is now well recognised that NbS offer significant potential to respond to global challenges, including converging climate change and urbanisation pressures. However, application globally remains uneven and fragmented (Li et al., 2021). Recognising this, IUCN has recently focused on identifying core NbS principles for successful implementation and upscaling; highlighting the importance of clarity of the evolution, definition, and key principles of NbS, as well as the links with related approaches (Cohen-Shacham et al., 2019). Central to developing this clarity has been devising evidence-based standards and guidelines to improve and increase the use of NbS interventions worldwide (ibid). IUCN's Global Standard for Nature-based Solutions was launched in 2020, aiming to provide a user-friendly and consistent framework for the verification, design, use and upscaling of NbS (International Union for Conservation of Nature (IUCN), 2020b). There are eight criteria and associated indicators in the Global Standard: 1) NbS effectively addresses key societal challenges (importantly, including ensuring that human wellbeing outcomes arising from NbS are identified and monitored); 2) design of NbS is informed by scale; 3) NbS result in net gains to biodiversity and ecosystem integrity; 4) NbS are economically viable; 5) governance mechanisms are appropriate; 6) trade-offs are balanced; 7) NbS are managing adaptively, from evidence; and 8) NbS are sustainable and "mainstreamed within an appropriate jurisdiction" (ibid). Also important in building a global best practice database on NbS is addressing the global inequities in documented NbS experience, including in urbanism (much focus is on Europe, as are the majority of researchers and authors) (Schröter et al., 2020; Li et al., 2021). This reinforces the importance of examination of NbS experience and findings from other areas of the globe, including Oceania.

EbA is typically thought of as a subset of NbS, specifically applying to the adaptation elements of climate change response, and aims to work with nature to adapt to climate change through strengthening biodiversity and ecosystems (Munang et al., 2013; Pedersen Zari et al., 2017). A key premise of EbA is that if ecosystems are protected, remediated or regenerated this will lead to healthier ecosystems, improved or increased ecosystem services, and thus enhanced human wellbeing and resilience to the impacts of climate change (Pedersen Zari et al., 2019). The unique nature of EbA is twofold: firstly, when considering ecosystem health, the provision of ecosystem services, and human wellbeing holistically, EbA can offer more participatory, flexible, and potentially more cost-effective solutions compared to '"hard"' engineered infrastructure adaptation strategies. Secondly, EbA approaches focus on, and reveal, multiple drivers of ecosystem change; including from *both* climatic changes and the activities of humans (Mackey et al., 2017; McPhearson et al., 2018). In this review we emphasise this potential of NbS to address climate change as well as urbanisation pressures.

NbS and EbA offer potential for both rural and urban areas. The potential of NbS for cities (at least in Europe and other developed nations) was given impetus in the mid-2010s by the European Commission's Horizon 2050 Expert Group on "*Nature-based Solutions and Re-naturing Cities*" (European Commission, 2015). This research and innovation agenda identified four, overlapping, principal goals that can be addressed by NbS: 1) enhancing sustainable urbanisation; 2) restoring degraded ecosystems; 3) climate change adaptation and mitigation; and 4) improving risk management and resilience (ibid). Seven NbS areas were recommended for prioritisation: (i) urban regeneration through NbS; 2) approaches closely linked to improved human wellbeing; 3) coastal resilience actions; 4) watershed management and ecosystem restoration; 5) NbS for increasing the sustainable use of matter and energy; 6) NbS for enhancing the insurance value of ecosystems; and 7) increasing carbon sequestration through NbS (ibid).

A large number of subsequent academic articles and studies from different disciplines have sought to advance the urban NbS agenda, largely with focus on European cities. Santiago Fink (2016), for example, highlighted the vital role of nature in addressing climate change at the city scale, focusing on green infrastructure as a cost-effective means to contribute to mitigation and adaptation priorities and simultaneously promote human wellbeing. Further, Frantzeskaki (2019) identified a number of key lessons for advancing NbS in European cities, including: 1) the importance of co-creation, and indeed citizen-led initiatives; 2) inclusive narratives and agendas; 3) a willingness and experiment and learn from innovation; 4) the critical role of collaborative governance as embraced by supportive local government; and 5) the importance of input from multiple disciplines and perspectives.

Dushkova and Haase (2020), focusing on urban design, point out that: 1) urban NbS projects have a much greater social, economic and environmental value than often originally understood; and 2) the co-benefits of NbS have the potential for great value when projects address the multiple needs of restoration, protection, and enhancement of ecological functionality and ecosystem services. Dushkova and haase identified five types of urban NbS approaches that could be applied to urban design: 1) NbS that make better use of protected or natural ecosystems in a way that increases urban ecosystem services supply; 2) NbS in conjunction with sustainable management of urban production systems such as urban forestry or farming; 3) NbS approaches that lead to the creation of new ecosystems (such as green walls, green roofs, and green buildings); 4) NbS approaches leading to the creation of new ecosystems from existing neglected, abandoned or brownfield sites; and 5) NbS associated with education and awareness on sustainable actions. These all present options for urban NbS for climate change adaptation and enhanced human wellbeing in Oceania, noting at present *urban* NbS experience is relatively limited in Oceania.

ECOSYSTEM SERVICES AND WELLBEING

As was highlighted and facilitated in terms of policy development by the United Nations' Millennium Ecosystem Assessment (MEA) of the mid-2000s, ecosystem services are fundamental to basic human survival and human wellbeing (Millennium Ecosystem Assessment (MEA), 2005). Ecologists have defined and categorised ecosystem services in various ways, but commonly within the four broad categories of: 1) provisioning services; 2) regulating services; 3) supporting services; and 4) cultural services. Recently, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) has developed as an influential alternative or complementary framework to that offered in the MEA. The IPBES approach emphasises human-nature relationships at the heart of an understanding of "nature's contribution to people" (Díaz et al., 2015; Pascual et al., 2017). The concept of ecosystem services is at the heart of both the MEA and IPBES models, but the framing and language are different. For example, the IPBES model highlights that ecosystem services are co-produced by social–ecological systems (Bennett et al., 2015). The IPBES model also acknowledges the bi-directionality between social and ecological systems. For example, human wellbeing can also influence institutional and governance provision of ecosystem goods and services (Leviston et al., 2018). The IPBES approach also highlights the contribution of ecosystem services to the Sustainable Development Goals (SDGs), the key international commitments of the 2030 Agenda for Sustainable Development (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), 2018). Notably, biodiversity protection is inherent in SDG 14 (conserve and sustainably use the oceans, seas and marine resources for sustainable development) and SDG 15 (protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss). In addition, there are potential contributions and options for policy makers, to ensure an understanding that ecosystem services contribute to all other SDGs. As just one example, focusing on SDG 3 (ensure healthy lives and promote wellbeing for all at all ages), there are clear, well established, links between healthy biodiversity and human health and wellbeing (ibid).

Overall, investigation into wellbeing has evolved across many disciplines including psychology, education, health, economics, ecology and geography among others; although there remains no universally-recognised definition or standard measurement of wellbeing (Pennock & Ura, 2011; Diener and Tov, 2012). The IPBES framework conceptualises wellbeing as comprising access to basic resources, freedom and choice, health and physical fitness, good social relationships, security, peace of mind, and spiritual experience. Wellbeing is considered achieved when individuals and communities can act meaningfully to pursue their goals and enjoy a good quality of life. The ecological connection is key, with living in harmony and balance with nature recognised as central to human wellbeing across cultures (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), 2021).

Understandings of wellbeing continue to evolve. It is our view that notions of wellbeing must be locally appropriate and nuanced to particular place-based and cultural circumstances in order to be useful. Given the unique region of Oceania (including its huge geographical span and diversity), increasingly pressured by climate change and urbanisation, an important question comes to the fore: "*how can we best conceptualise wellbeing within an ecosystem services approach* *to urban development in the Pacific?"* We return to this question in later sections.

PACIFIC ISLAND CASE-STUDIES

Context

Kiribati, Samoa and Vanuatu, amid similarities and differences, showcase the challenges brought by coalescing climate change and urbanisation pressures for Pacific small island nations and also other small island developing states (SIDS) globally. All three Pacific nations have become leading regional and international voices calling for increased global attention to climate change mitigation and adaptation and biodiversity challenges. The capital cities of these nations, described in the following paragraphs, although small in global terms, heavily dominate the population and economy of the islands on which they are situated. More significantly, these cities have all experienced rapid population growth rates in recent decades, much higher than their countries' national and rural growth rates. Population growth has occurred not only within urban administrative boundaries but also in peri-urban peripheries, affecting not only the population distribution of the urbanised areas, but also the provision of infrastructure and social services, food security and the cultural and social institutions of their settings.

Kiribati, in Micronesia, is a nation of 33 islands and approximately 118,000 people living in 21 islands (20 coral atolls and one volcanic island) spread across approximately 3.5 million square kilometres of ocean. Total land area is only 810 square kilometres. In 2019, 54% of the population was urban (World Bank, 2021), with South Tarawa the largest urban centre (population approximately 56,000), regularly reported as one of the densest urban agglomerations in the Pacific, if not the world. Kiribati's population continues to rapidly urbanise (2.9% urban growth in 2019), shaped by movement from outer islands to South Tarawa in particular (ibid). Kiribati's coral atolls only reach a few metres above sea level (on South Tarawa, for example, the highest elevation is 3 m). 35% of the population live within 0–50 m of the ocean, with a further 52% within the 50–100 m band (Kumar et al., 2020). Significant issues for Kiribati, among others, include the potentially existential impacts of sea level rise and increased storm events from climate change, limited freshwater and the salinification of freshwater lenses, and the need for improved sanitation and solid waste management. The incidence of basic needs poverty is highest in urban South Tarawa, affecting 24.2% of the population (Government of Kiribati, 2016a). Overall, Kiribati is confronted by a perfect storm of inherent climate vulnerability, limited land, continuing urbanisation, and overcrowding (Cauchi et al., 2019).

Samoa, in Polynesia, has a population of approximately 198,000 people spread across the two large volcanic islands of Upolu and Savai'i and eight smaller islands. The urban population was 18% in 2019 (World Bank, 2021), with the capital, Apia, the largest urban centre (approximately 36,700 people). Approximately 70% of the population reside in 330 villages along the coasts of Upolu and Savai'i (Government of Samoa, 2013). The natural hazard risk profile, impacted by climate change, is significant–with the country experiencing a number of devastating disasters in recent times, including cyclones and tsunamis as the most damaging. The National Climate Change Policy for Samoa highlights the significant sustainable development challenge: "Samoa shares with other SIDS the characteristics of being economically vulnerable and ecologically fragile because of its geographical location, isolation, limited resources and exposure to global economic crisis. Climate change impacts are [also] an added imposition on the inherent challenges Samoa already faces" (Government of Samoa, 2020).

Vanuatu, in Melanesia, has a population of approximately 282,000 people spread across a large volcanic archipelago of 83 islands. Like other nations of Melanesia, there is huge cultural and linguistic diversity. More than one hundred indigenous languages are spoken, for example. The nation was 25% urban in 2019, with urban growth globally-high at 2.9% in 2019 (World Bank, 2021), and urban and peri-urban growth rates in key administrative divisions reaching more than 8% between 1999 and 2009 (Trundle & McEvoy, 2015). The capital Port Vila, on the island of Efate, is the largest urban centre, with a population of approximately 51,500. Around 43.8% of the population lives within 500 m of the ocean (Kumar et al., 2020). Vanuatu ranks extremely highly in various global natural hazards vulnerability indices. It is exposed to cyclones and other storm events, earthquakes, and volcanic activity in particular. More than half the population, for example, are impacted annually by climate related extreme events or geohazards (Radtke et al., 2018).

Adaptation Priorities

Pacific island nations are insignificant emitters of greenhouse gases. Kiribati's Intended Nationally Determined Contribution document, for example, highlights that the nation's emissions per capita are among the lowest globally (Government of Kiribati, 2016a). As such, the focus of government efforts to respond to climate change in Kiribati, Samoa and Vanuatu has been on adaptation initiatives in various forms, often supported by a large number of partners, including regional organisations, multilateral organisations through various global funds, and bilateral aid programmes.

Climate change adaptation features prominently in high-level policy documentation in all three case-study nations, amongst advocacy for global mitigation efforts. A key aspiration of *Vanuatu 2030: The People's Plan* is "enhanced resilience and adaptive capacity to climate change and natural disasters" (Government of Vanuatu, 2016a). *The Vanuatu Climate Change and Disaster Risk Reduction Policy 2016–2030* provides more specificity on priority actions, bringing focus on the areas of disaster risk reduction (DRR), community-based adaptation (CbA), and ecosystem-based approaches (Government of Vanuatu & Pacific Community, 2015). Similarly, the *Kiribati 20-Year Vision 2016–2036* identifies environment and climate change as a key cross-cutting issue and highlights the critical need to mainstream climate change adaptation and mitigation across government policy and programmes (Government of Kiribati, 2016b). Providing more detail, the recent *Kiribati Development Plan 2016–2019* identifies "environment" as one of six priority areas with an associated goal to "facilitate sustainable development through approaches that protect biodiversity and support the reduction of environmental degradation as well as adapting to and mitigating the effects of climate change" (Government of Kiribati, 2016c).

In Samoa, the recent *Strategy for the Development of Samoa 2016–2020* (SDS) premised four priority areas including "environment" (including the key outcome areas of "environmental resilience improved" and "climate and disaster resilience increased") (Government of Samoa, 2016a). It is notable, however, that the more recent *Samoa 2040: Transforming Samoa to a Higher Growth Path* policy document that complements the SDS does not specify environment or climate-related priorities beyond investment in "climate and disaster resilient infrastructure" (Government of Samoa, 2021).

Have adaptation efforts been successful in Kiribati, Samoa and Vanuatu? Academic literature investigating this question is relatively limited and, overall, the picture is mixed. Webber (2015), for example, investigated the significant World Bank-funded two-phased Kiribati Adaptation Project (KAPI and KAPII) and highlighted that while both focused on hard infrastructure (especially the construction of seawalls), it was ecosystem-based aspects, notably mangrove rehabilitation and planting, that were the more successful elements of both projects (as assessed in formal project evaluations).

More focus in the academic literature has been given to evaluating the success or otherwise of CbA projects, generally critiquing efforts to date. Piggott-McKellar et al. (2020), for example, report on the evaluation of a rural CbA project in Abaiang Island in Kiribati, concluding that outcomes were largely ineffective and unsustained. They highlight the key lesson that local contextual factors such as social norms, environmental, or local governance and decision-making mechanisms must be identified and meaningfully incorporated into the design and implementation of CbA initiatives. Similarly, Cauchi et al. (2021), acknowledging the top-down nature of many adaptation projects, highlighted through a series of participatory focus groups in Kiribati, how critical it is to ensure communities participate in the co-design of adaptation interventions. These CbA findings resonate with research from Samoa that has highlighted that to understand climate change resilience in an island society, careful assessment of islanders' perceptions and actions in the context for their physical locales and socio-cultural systems is required (Latai-Niusulu, 2016). In short, islanders have detailed understanding, awareness and experience of climate changes (ibid) and this knowledge is vital to incorporate into adaptation initiatives.

Evaluations in Vanuatu have also reported the challenges of CbA initiatives to date. Westoby et al. (2020), for example, reviewed research evaluating 15 CbA projects in Vanuatu and concluded they invariably fell short of success, longevity, and sustainability. They argued that CbA projects typically were led by external "experts" working temporarily in local communities in sporadic design and implementation stages, "fitting" efforts to funding requirements and failing to view local communities as best placed to define and shape resiliency agendas. They concluded that localised adaptation efforts must be locally led and implemented across different entry points, and not just necessarily related to individual specific "communities".

Overall, contextual specificities are vital to understand and incorporate in adaptation efforts (Clarke et al., 2019). This is also essential for adaptation and climate resilience in urban areas. Trundle (2020), for example, through case-studies of environment- and climate-vulnerable informal settlement communities in the capital cities of Port Vila (Vanuatu) and Honiara (Solomon Islands), shows how important sub-city analysis provides detail on urban resilience strategies such as informal maintenance of ecosystem services, use of kinship and familial networks, and the translocation of traditional knowledge.

NbS and EbA Approaches

Overall, there are many projects currently operating in the Pacific islands region that are broadly classifiable as NbS and/or EbA. Some are regional initiatives, and some specific to an individual Pacific island nation or territory. A 2019 review commissioned by the New Zealand Ministry of Foreign Affairs and Trade, for example, identified 31 projects aimed at delivering resilient ecosystem services under the broad heading of NbS. The majority of these projects focussed on adaptation to climate change through awareness, conservation, restoration, and sustainable management of natural resources (Douglas et al., 2019). Geographically, projects were focused across a number of different scales or continua, such as: between rural and urban, high volcanic islands and low atolls, and main and outer islands. Eight of the projects (in Samoa, Vanuatu, Solomon Islands, Fiji, and Marshall Islands) were identified as having a specific focus on urban areas (ibid). On the other hand, a policy review conducted for UNESCAP (UNESCAP, 2019) identified a lack of effective urban governance structures and mandate, and weak or fragmented local and national government structures for urban management as significant barriers to implementation of urban NbS. The policy recommendations of this review included measures aimed at elevating a blue urban agenda in responsible levels of government at the local, provincial and national levels.

Regionally, Vanuatu has been a leader in EbA approaches, with EbA featuring prominently in key government policy documentation. For example, the *2016–2030 Climate Change and Disaster Risk Reduction Policy* identifies targeted EbA actions including "ridge to reef" solutions, prioritising "soft" interventions such as coastal revegetation (compared to "hard" engineered infrastructure such as seawalls), advocacy and awareness programmes, and activities that build on existing local "taboos, conservation areas, heritage sites, locally managed areas and vulnerable habitats and ecosystems and carbon sinks" (Government of Vanuatu & Pacific Community, 2015). Notably, the policy also brings considerable focus to the role of TEK into adaptation planning, design and implementation, while also noting the importance of including TEK into formal and informal school curricula (ibid). Further, the *National Ocean Policy* highlights an ecosystem-based approach as the foundation of ocean management while also acknowledging the important role of TEK (Government of Vanuatu, 2016b). The Government of Vanuatu's focus on incorporating TEK comes from the strongly held and widespread conviction that the traditional economy is vital for subsistence, livelihoods, and wellbeing in Vanuatu (Regenvanu, 2010; Government of Vanuatu, 2016a) and that happiness (or subjective wellbeing) is inherently linked to access to customary land and natural resources, traditional knowledge and practice, and community vitality (Malvatumauri National Council of Chiefs, 2012).

A significant EbA project active recently in Vanuatu (and also Fiji and Solomon Islands) is the Pacific Ecosystems-based Adaptation to Climate Change (PEBACC) project, the first stage implemented by SPREP from 2015 to 2020 with funding from the German Government. PEBACC involved four key stages: 1) ecosystem and socio-economic resilience analysis and mapping; 2) EbA options assessments; 3) development of EbA implementation plans; and then 4) implementation of pilot projects (SPREP, 2020). The PEBACC project included a specifically urban focus in Vanuatu (Port Vila) and Solomon Islands (Honiara).

In Vanuatu, PEBACC evaluation focussed not only on the officially recognised urban area of Port Vila, but also its surrounding peri-urban area and the large water catchment within which both these areas are located, in a ridge-to-reef approach, acknowledging that the terrestrial, freshwater, and coastal ecosystems of small islands are highly interconnected (Pedersen Zari et al., 2020). Application of the first three PEBACC stages identified above resulted in the identification of five EbA priorities: riparian corridor regeneration; restoration and protection of coastal vegetation; intensification of home gardens; urban tree planting; and the use of traditional housing technology in a demonstration sustainable urban housing project. The use of the PEBACC methodology in Port Vila provided a number of important lessons: 1) the needs of local communities must be at the forefront of project planning, requiring a participatory design process; 2) EbA project development must be multidisciplinary and iterative; 3) appropriate data, both quantitative and qualitative, are vital as a basis for EbA project development, and adequate time for data gathering is required; 4) urban and coastal EbA projects must be developed holistically, recognising socio-ecological systems that extend beyond urban areas; 5) the complex overlapping landscape of governmental and international aid financed projects must inform the development of new EbA projects; 6) potential monetary and non-monetary benefits, costs and risks across multiple factors must be carefully assessed; and 7) project implementation requires ongoing engagement and a readiness to adapt to on-the-ground realities that may shift (Pedersen Zari et al., 2020).

NbS/EbA activities also feature highly in the suite of activities that Samoa has prioritised in its climate change adaptation efforts. Chong (2014) notes, for example, that "EbA is well integrated within five of the nine priority projects identified in the NAPA [National Adaptation Programme of Action], which makes explicit the value of ecosystem services to building the adaptive capacity of communities". Within the urban context, the most significant adaptation project incorporating NbS/EbA elements is the US$65 million Global Environment Facility (GEF)-funded and United Nations Development Programme (UNDP) implemented Vaisigano Catchment Project (VCP). The overall purpose of VCP is to strengthen adaptive capacity and reduce exposure to climate risks faced by communities and infrastructure in the catchment area of Apia (Green Climate Fund, 2021a). The project includes significant hard infrastructure components but also includes ecosystem responses such as crop planting, Ecosystem-based Adaptation Enterprise Development (EbAED) (supporting small businesses to engage in activities that will improve ecosystem function and have climate change adaptation benefits), cash for work through green jobs, and payment for ecosystem services (PES) (Green Climate Fund, 2016; Douglas et al., 2019). To date, some 20 EbAED projects are in operation with a further 319 projects approved; cash for work schemes are underway for ecological rehabilitation programmes at three reserve sites and fencing for watershed protection at one further site; and the PES component continues through feasibility stages (Samoa Ministry of Natural Resources and Environment, 2021).

In Kiribati, a number of projects, implemented at various scales (but particularly in rural areas), are broadly classifiable as NbS/EbA interventions, or include NbS/EbA components. The ongoing Food and Agriculture Organization (FAO)-implemented "Resilient Islands, Resilient Communities" project, for example, aims to improve biodiversity conservation and landscape and seascape level management to enhance socio-environmental resilience to climate variability and change. Funded by GEF (US$18 million), the project focuses on ridge-to-reef approaches for food security, sustainable livelihoods, and restoration and conservation of natural resources. Secondly, the UNDP-implemented "Enhancing National Food Security in the Context of Climate Change" (2016–2020) project looked to improve food security and hence the adaptive capacity of vulnerable communities through activities seeking to enhance ecosystem integrity such as coral reef restoration and improved ecosystem management (Douglas et al., 2019).

Opportunities

Climate change adaptation is an absolute priority for Kiribati, Samoa and Vanuatu, as well as other Pacific island nations. But adaptation is difficult. As evaluations of CbA projects have shown, for example, success is not guaranteed. NbS/EbA offer considerable potential for putting healthy ecosystems and biodiversity, crucial ecosystem services, and the key link between healthy ecosystems and human wellbeing at the centre of adaptation efforts, including in and particularly for urban areas. In Oceania, as elsewhere, NbS and EbA present opportunities for cost effective approaches, hybrid solutions, and the support of livelihoods through the restoration, regeneration, and protection of terrestrial and marine natural resources. Critically, NbS and EbA also offer significant opportunity for incorporating TEK, so rich in the region, into adaptation efforts.

Many projects and partners are already active in NbS/EbA in Oceania. But much more can be done. The New Zealand Ministry of Foreign Affairs and Trade-commissioned review of NbS in Oceania, for example, recommended three broad categories of NbS opportunities that could fill current gaps: 1) restoring traditional gardening and farming practices, where eroded, for ecosystem health, food security, and improved human health benefits; 2) the prospects of traditional food storage methods to support disaster preparedness; and 3) the potential for using bio-indicators as early warning systems for climatic events such as droughts (Douglas et al., 2019). Recognising that significant gaps in implementation persist, a major 35 million euro multi-donor NbS programme was launched in 2020, led by Agence Française de Développement (with support from SPC, SPREP and IUCN), called the Kiwa Initiative. This programme aims to strengthen climate change resilience for Pacific island ecosystems, communities, and economies through NbS that protect, sustainably manage, and restore biodiversity (Pacific Community, 2021a). The Kiwa Initiative will provide grants for a variety of local and regional projects and provide associated technical assistance for project proposal development. These projects are likely to focus on a variety of different geographic scales, including both rural and urban areas. The Kiwa Initiative explicitly puts "people at the heart of its priorities [to] help drive forward socially inclusive project implementation at all levels" (Pacific Community, 2021b)—recognising that those most impacted by climate change, and depending the most on natural resources for their livelihoods, are best placed to develop and implement long lasting NbS (ibid). Future phases of PEBACC in Fiji, Vanuatu and Solomon Islands, focused on the implementation of EbA projects developed in the first phase, are also currently planned to be funded via the Kiwa Initiative. Some are likely to have an urban focus, such as those discussed already planned for Port Vila. Another major initiative in preparation is the Green Climate Fund Melanesian Coastal and Marine Ecosystem Resilience Programme (M-CMERP). This project focused on Papua New Guinea, Solomon Islands, and Vanuatu will look to prioritise and integrate EbA in national planning and decision-making amid long-term (50–30 years) climate impact and resilience scenarios, as well as provide grants to EbA and resilient development investments (Green Climate Fund, 2021b).

Opportunities also exist, or may present, through various government priorities and flagship projects for NbS/EbA in urban areas. In Samoa, for example, the National Adaptation Programme of Action (NAPA) specified zoning and strategic urban management adaptation priorities aiming, *inter alia*, for environmental dividends by strengthening adaptive capacity and urban intensification through an improved urban centre from the promotion of attractive design and heritage (Government of Samoa, UNDP and GEF, 2005). This helped lead to the creation of the Planning and Urban Management Authority (PUMA), responsible for managing Apia's urban growth, and recent work revitalising Apia's waterfront aiming to improve attractiveness, functionality and safety (Government of Samoa, 2016b). Key elements of the Apia waterfront work involve the protection of green spaces, parks, reserves, streetscapes and other recreation spaces (ibid); fertile ground for the potential application of NbS/EbA approaches. In addition, in Kiribati, considerable effort and funds have been directed at planning and feasibility stages of the Temaiku Land and Urban Development project aimed at reclaiming and raising (by 2–5 m) 300 ha of land on South Tarawa to provide a "resilient basis for future land and urban development [with] the potential to house 35,000 people" (Watkin et al., 2019). This project, likely enormously costly and still uncertain, was planned to combine phased hard and soft coastal defence solutions and a range of uses including residential housing, government buildings, infrastructure and utilities, and recreation. NbS/EbA approaches, if prioritised, could be incorporated into this project should it be realised.

As discussed, ecosystem services and their connection to human wellbeing and survival, are central to NbS and EbA approaches. In urban areas, adaptation approaches that premise wellbeing offer great potential. In Vanuatu, for example, where the government has a strong interest in wellbeing and its determinants, research has shown that subjective wellbeing, or happiness, is lower, on average, in urban areas compared to rural areas (Malvatumauri National Council of Chiefs, 2012). As introduced earlier, the same research has highlighted how wellbeing in Vanuatu is linked to three key factors: 1) access to customary land and natural resources; 2) traditional knowledge and practice; and 3) community vitality. Thus, in Vanuatu and likely elsewhere, it is clear that NbS and EbA approaches for climate change adaptation; which work *with* nature at their very core, offer great potential for improving urban wellbeing, particularly when combined with approaches that are driven by or incorporate TEK.

CONCLUSION

Across both rural and urban areas there are critical linkages between ecosystems, ecosystem services, and human health and wellbeing. In the Pacific islands region climate change and urbanisation combine to profoundly impact ecosystems, ecosystem services, and the livelihoods that they support. As described by Cauchi et al. (2021) and Pedersen Zari et al. (2019), climate change can be seen as a multiplier of urbanisation and other environmental pressures. Adaptation to climate change is urgently required, and NbS and EbA approaches offer great potential across different scales. Our three Pacific island case studies showcase the growing evidence base of NbS and EbA approaches in Oceania. But gaps in knowledge, policy, and practice remain, particularly for rapidly growing urban and peri-urban areas. It is also clear that successful adaptation requires careful consideration of the local context and participatory "bottom-up" co-design and implementation with local communities (Kabisch et al., 2017; Narayan et al., 2020; Piggott-McKellar et al., 2020; Cauchi et al., 2021). We believe that there is great potential for a nature-based urban design agenda positioned within an urban ecosystems framework linked closely to Indigenous, localised, understandings of wellbeing and ecology. The co-design and implementation of urban NbS would be the defining features of this agenda, building from key lessons elsewhere that local communities must be inherently involved in NbS planning, design, and implementation (Kabisch et al., 2017; Frantzeskaki, 2019; Dushkova & Haase, 2020; Li et al., 2021; Seddon et al., 2021). Building such an agenda is an important contribution to nature-based ecological urban design in Oceania, particularly given that spatially explicit urban design policy and practice is often absent in Pacific islands nations.

We posed the question earlier "'*how can we best conceptualise wellbeing within an ecosystem services approach to urban development in the Pacific?*" As yet, the answers to this question are still far from clear. But we suggest that progressing an Oceania urban NbS agenda and responding to this question requires: 1) developing an inventory of innovative urban NbS strategies for the region; 2) more comprehensively exploring the range of existing and potential Indigenous wellbeing frameworks within Oceania; 3) using community co-design to develop future urban NbS strategies centred in TEK and related Indigenous wellbeing frameworks; and 4) ensuring that the wellbeing of Indigenous peoples, however defined locally, is a central pillar of future Oceania urban design and climate change adaptation initiatives.

Indigenous knowledges have long held that human wellbeing is inextricably connected to ecosystem health. We believe that building on Indigenous framings of wellbeing, and partnering TEK and other scientific information with NbS, can lead to place-based, localised, design responses that can offer long-term benefits across different scales, including in urban areas. Further developing an Oceanic urban design agenda is the focus of ongoing research undertaken by a collaboration of Aotearoa New Zealander, I-Kiribati, Samoan, and Ni-Vanuatu researchers and practitioners, including the authors. The recently released sixth assessment report (2021) of the Intergovernmental Panel on Climate Change (IPCC) has highlighted the acute vulnerability of Pacific island nations to climate change (SPREP, 2021). Adaptation efforts are vital, including in urban areas that are instrumental in contributing to global climate and sustainability goals (Santiago Fink, 2016; Li et al., 2021). Urban design responses, including those working with nature and with community co-creation at the core, will be an integral part of efforts to adapt in ways that protect and enhance the wellbeing of people and the ecologies of the region.

AUTHOR CONTRIBUTIONS

GK lead authorship. TB, AL-N, WM, MPZ, RK, VC, PB, DL equal supporting input.

FUNDING

We would like to acknowledge funding from the Marsden Fund – Royal Society of New Zealand and from two Victoria University of Wellington Faculty of Architecture and Design Innovation strategic research grants.

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frontiers in Environmental Science

ORIGINAL RESEARCH
published: 04 November 2021
doi: 10.3389/fenvs.2021.679189

Dryland Watershed Restoration With Rock Detention Structures: A Nature-based Solution to Mitigate Drought, Erosion, Flooding, and Atmospheric Carbon

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OPEN ACCESS

Edited by:
Borja Gonzalez Reguero, University of California, Santa Cruz, United States

Reviewed by:
Peter William Clinton, New Zealand Forest Research Institute Limited (Scion), New Zealand
Thomas Sisk, Northern Arizona University, United States

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Specialty section:
This article was submitted to Interdisciplinary Climate Studies, a section of the journal Frontiers in Environmental Science

Received: 11 March 2021
Accepted: 18 October 2021
Published: 04 November 2021

Citation:
Gooden J and Pritzlaff R (2021) Dryland Watershed Restoration With Rock Detention Structures: A Nature-based Solution to Mitigate Drought, Erosion, Flooding, and Atmospheric Carbon. Front. Environ. Sci. 9:679189. doi: 10.3389/fenvs.2021.679189

Historic land degradation is an ongoing threat to the Sky Islands of southern Arizona, US, and northern Sonora, Mexico, an area designated as a globally significant biodiversity hotspot. Land degradation has reduced ecosystem services provisioning, released carbon from disturbed soils into the atmosphere, and significantly diminished resilience to climate change. Private land managers in the region have developed methods to reverse degradation and restore biodiversity and ecosystem function. Land managers have used rock detention structures (RDS), technology adapted from traditional Indigenous practices in the region, as a tool for reversing desertification and watershed degradation. The structures were installed primarily for erosion control and water management, but they have had positive impacts on multiple biophysical systems. In this study, we analyze watershed-scale installation of RDS as a nature-based solution for climate change mitigation and adaptation. Case studies include four properties that offer examples of structures that have been in place over a period ranging from 1 to 40 years. We reviewed journal articles and other studies conducted at the four sites, supplemented with interviews, to catalogue the nature-based solutions provided by RDS. This study documents positive impacts on overall stream flow, reduction in peak runoff during inundation events, and increased sedimentation, which increase resilience to drought, erosion, and flooding. Data suggest potential impacts for climate change mitigation, though further research is needed. In addition, results suggest that watershed restoration with RDS offers a host of co-benefits, including an increase in biodiversity and wildlife abundance, an increase in vegetative cover, and increased surface water provisioning over time to support the land-based livelihoods of downstream neighbors. In the discussion, we consider barriers to replication and scalability using the strategy of the UN Decade on Ecosystem Restoration as a guiding framework, discussing issues of awareness, legislation and policy, technical capacity, finance, and gaps in knowledge.

Keywords: natural climate solution, ecosystem services (ES), erosion control structure, riverine, wetlands, carbon sequestration, carbon market, conservation finance

INTRODUCTION

Ecosystems of the arid southwestern United States and northern Mexico have suffered extensive land degradation. This degradation has been a continuing threat to the Sky Islands of southern Arizona and northern Sonora, an area that is both a biodiversity hotspot and a harbinger of the deleterious effects of climate change (Deyo et al., 2012; Falk, 2013).

One of the most visible symptoms of land degradation in the Sky Islands has been the incision of streambeds and the declining health of the riparian areas that depend upon them. Though the region is arid, it was crossed with important rivers and wetlands (ciénegas) at the time of European arrival (Minckley et al., 2009). Even in a degraded state, these perennial and ephemeral waterways are critical to maintaining the region's biodiversity. For example, in the Sonoran Desert, dry washes occupy less than 5% of land area, but they support 90% of bird life, even though washes may carry water for only a few hours a year (Dimmit 2015).

Healthy riparian areas play important roles in regulating the flow and quality of surface water, providing water for domesticated animals and wildlife, maintaining water tables, recharging aquifers, and preventing erosion. Globally, wetlands, including riparian areas, constitute only 9% of landscapes, but they are estimated to deliver 23% of global ecosystem service values (Zedler and Kercher 2005; Costanza et al., 2014). When watersheds are degraded, the provision of these ecosystem services is diminished. Degraded watersheds are characterized by disturbances to hydrology brought about by poor soil stability due to a lack of vegetation, unsustainable timber harvest, over-allocation of surface and groundwater, overgrazing, intentional burning for agriculture, a legacy of fire suppression, habitat fragmentation, and extirpation of beaver (*Castor canadensis*, Cole and Cole 2015).

In recent years, scientists and policy makers have begun addressing loss of ecosystem functionality through nature-based solutions (NbS), which are "actions to protect, sustainably manage, and restore natural or modified ecosystems that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits" (IUCN 2020). Recognized societal challenges that can be ameliorated through NbS include climate change mitigation and adaptation, disaster risk reduction, economic and social development, human health, food security, water security, environmental degradation, and biodiversity loss (IUCN 2020).

NbS have the potential to remedy societal problems that have traditionally been addressed with grey infrastructure. Unlike grey infrastructure, however, NbS often deliver a host of co-benefits (Chausson et al., 2020). For example, on vulnerable coastlines, both dykes and restored mangroves can address erosion and storm surges, but restored mangroves also benefit biodiversity, carbon sequestration, and food security (McGinn 2019). Research has found that most NbS have positive outcomes, such as increased number of species, functional diversity, or greater plant or animal productivity (Chausson et al., 2020). NbS can also foster a sense of place and strengthen social infrastructure (Tidball et al., 2018). In this study, we use NbS as a lens for examining the impacts of watershed restoration on ecosystem functionality.

Indigenous peoples in this region have installed rock detention structures (RDS, a subset of the more general category of erosion control structures) for centuries, most famously (in this region) around Trincheras, Sonora (Phillips, 2009). The archaeological literature on the American Southwest contains thousands of records of check dams, and the Zuni people of the Southwest continue to use check dams to spread the flow of water more evenly over the land, as they and others have for centuries (Doolittle, 2010).

Most recently, restoration practitioners have been installing RDS at our case study sites (see *Methods*) and elsewhere to increase water availability, reduce the number and severity of flooding events, promote vegetative growth, and decrease erosion. RDS are simple structures, usually composed of stone found on site, and they include a variety of types, such as check dams, one rock dams, and gabions **Figure 1**. When the structures installed are of small or medium size, such as those illustrated in **Figure 1**, RDS-based watershed restoration is an example of low-tech, process-based restoration (Wheaton et al., 2019), which is defined as the use of simple, cost effective, hand-built solutions that help repair degraded streams and that are "designed to kickstart the processes that allow the stream to repair itself" (Wheaton et al., 2019, p.4). While their construction is an act of craftsmanship, and proper placement requires expertise, volunteers can be trained to construct high quality structures. Their impact can be profound: "Viewed in terms of environmental change, there may well be no deliberate human activity that has a greater impact given its minimal input than the construction of check dams. The simple act of dropping an obstacle across a small water course affects land both upstream and downstream" (Doolittle, 2010).

The "low tech" aspects of this solution make it worthy of further investigation for several reasons. First, globally more than 2 billion hectares of previously productive land is now degraded (UNCCD, 2020), and much of that land is occupied by the world's poor. Relative to "high tech," engineered alternatives, solutions with low-cost materials and minimal training can be more rapidly deployed and used to better engage communities in land stewardship. Additionally, in many regions of the world, people already use RDS for erosion control and to improve land for agriculture (WOCAT, 2021). Documenting the additional utility of watershed-scale installation extends their applicability to restoring other ecosystem functions. Lastly, because RDS slow and spread water, their use in arid and semi-arid regions may help foster resilience to current and future challenges associated with a changing climate.

The aims of this paper are to identify 1) the multiple societal impacts of watershed-scale installation of RDS in order to understand their function as an NbS and 2) factors that affect replicability and scalability in the Sky Islands. To address the first aim, we consolidate empirical evidence from case study sites to catalogue the societal problems for which RDS can serve as a solution. We investigated four case studies in northern Sonora and southern Arizona that offer examples of RDS that have been in place over a period ranging from 1 to 40 years. Using both interview data and analyses from 18 journal articles, theses, and other publications based on data collected at these sites, we document the impacts on overall stream flow, peak flow during inundation events,

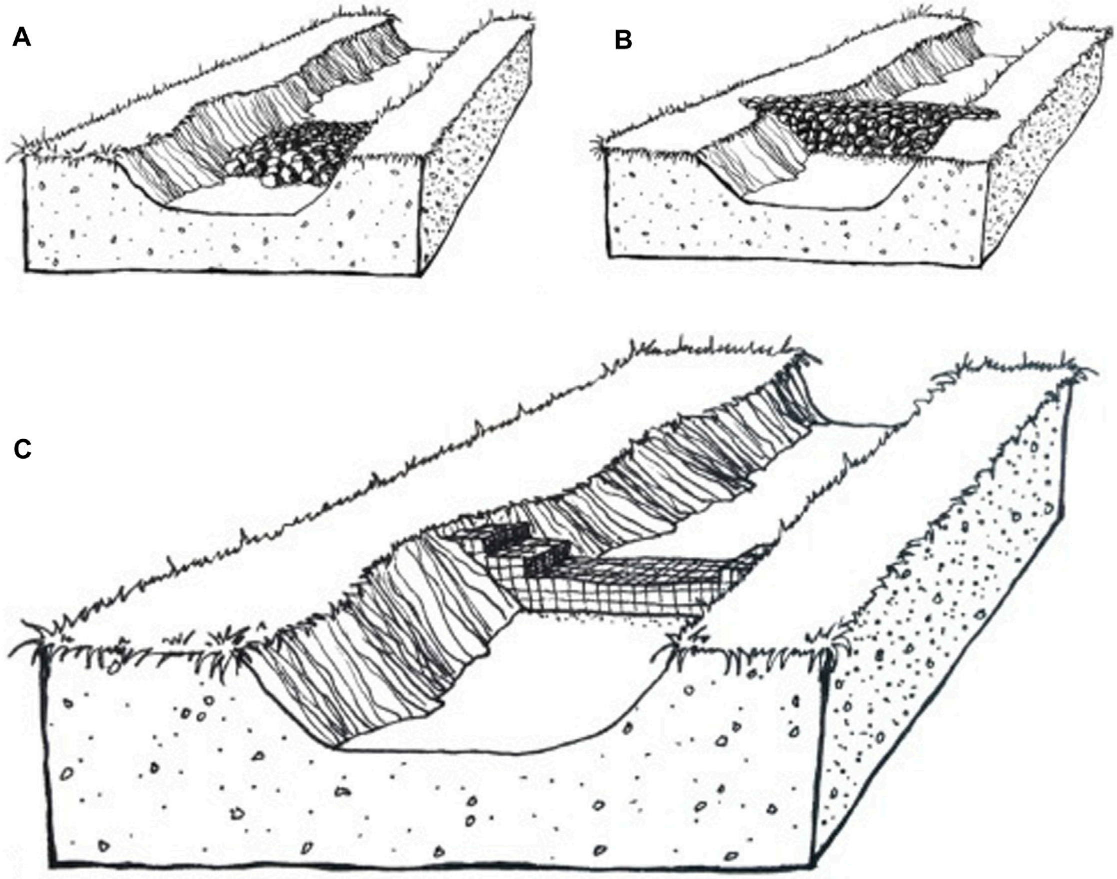

FIGURE 1 | Sketches of rock-detention structures, including: **(A)** spreader (or one-rock dam), **(B)** loose-rock check dams (or gully plugs), and **(C)** larger rock-filled wire baskets (gabions) (reprinted from Norman et al., (2017), **Figure 1**, Chloé Fandel).

sedimentation and the resulting effects on resilience to drought, erosion, and flooding. In addition, data suggest that watershed restoration with RDS offers a host of other co-benefits, including an increase in biodiversity and wildlife abundance, an increase in vegetative cover, and increased surface water provisioning over time to support the land-based livelihoods of downstream neighbors.

To address the second aim, we consider barriers to widespread implementation using the strategy for the United Nations Decade on Ecosystem Restoration as a guiding framework, informed by our experience with watershed restoration in the Sky Islands. We consider how barriers can be overcome, including guiding principles to inform policy decisions and the potential of carbon markets as a financing mechanism.

METHODS

Sky Islands Ecosystem

All four case studies are located within the Sky Islands/Madrean Archipelago in southeastern Arizona and northeastern Sonora. This region lies at the convergence of several major biogeographic zones, including the Sonoran Desert, Chihuahuan Desert, the Rocky Mountains and Colorado Plateau, and the Sierra Madre Occidental. It is home to more than 60 isolated mountain ranges with elevations that rise to over 3,000 m.

The Sky Islands sustain exceptional biodiversity. For example, approximately one-third of all bird species in North America occur in the Chiricahua Mountains, an isolated range in southeastern Arizona (Moore 2015), and one-quarter of all species native to Mexico occur in the Madrean pine-oak woodlands (Wilson, 2016). The entire region is thought to be home to 3,600 species of plants (Moore, 2015). The Sonoran portion is less well known, but new species and genera continue to be discovered in ongoing explorations.

In Tucson, AZ, temperatures increased 0.25C per decade between 1949 and 2011 (Moore, 2015). As a consequence, there have been more large fires, widespread outbreaks of bark beetles, spread of invasive species, shifts in plant flowering times, and plant species being pushed to higher elevation, potentially causing local extinctions of high-elevation endemics and other evolutionary unique lineages (Moore, 2015).

The Sky Islands have been designated an urgent conservation priority due to the region's high biodiversity and high level of threat. In the last 200 years, the Sky Islands have been degraded due to the pressures of unsustainable grazing, farming, mining, and eradication of keystone species, such as beaver (*Castor canadensis*, Cole and Cole, 2015). Ciénegas, or desert wetlands, and meandering waterways were drained by the historical incision of rivers (Hendrickson and Minckley, 1985), which depleted surface and ground water, altered aquatic habitats, and led to a decline in the wildlife that depend on them (Cole and Cole, 2015). Conservation International, the Critical Ecosystem Partnership Fund, and the Key Biodiversity Areas Partnership have designated the area a global biodiversity hotspot, highlighting its plant diversity and threat of further degradation (Critical Ecosystem Partnership Fund, 2020; Conservation International, 2020; Birdlife International, 2020).

Case Study Sites

In this paper we analyze evidence for the impacts of RDS at four case study sites with which we are familiar. The sites are the Cuenca Los Ojos protected area in northeastern Sonora, Mexico, and the El Coronado Ranch, Smith Canyon, and the Babacomari Ranch in southeastern Arizona, United States.

Cuenca Los Ojos

Cuenca Los Ojos (CLO) is a nonprofit privately protected area in northern Sonora, Mexico, immediately south of the U.S.-Mexico border. The 53,000-ha area consists of nine former cattle ranches, ranging in elevation from 900 to 2,400 m. Low elevation desert grasslands transition to mesquite grasslands and pinyon-juniper woodlands, and upper elevations support dense coniferous forests. CLO lands provide habitat for at least 13 endangered or threatened species. Species counts include more than 70 mammals, 250 birds, 20 amphibians, 70 reptiles, 8 native fish (4 endemic), 900 invertebrates, and an estimated 3,000 plants (pers. comm.).

Prior land uses, including grazing, significantly degraded the land, accelerated erosion and attendant release of carbon, and reduced water infiltration, which depleted water tables. Restoration efforts have included changes to grazing practices and installation of RDS in eroded areas, beginning in the mid-1990s. To date, CLO has constructed more than 50 large gabions and 40,000 small rock dams in lower order washes and hillsides. It is one of the largest-scale applications of this method known; DeLong and Henderson, (2012) noted, “we are unaware of a comparable attempt to use gabions and berms for the sole purpose of ecological restoration along >10 km of arroyo channels draining watersheds on the order of ~400 km^2 and larger.” Current threats include climate change, adjacent land use change to irrigated cropland, and habitat fragmentation by both the U.S.-Mexico border wall and the widening of Highway 2 in Mexico.

El Coronado

El Coronado (EC) is a 6,600-ha ranch on the western slope of the Chiricahua mountains in southeastern Arizona, including both privately owned land and the leased West Turkey Creek allotment from the U.S. Forest Service. The proprietors began installation of RDS structures in the Turkey Pen watershed in 1983 to mitigate erosion caused by previous cattle grazing and forest clearing, eventually installing approximately 2,000 structures. After successful demonstration of watershed restoration at El Coronado, proprietors founded CLO to extend their impact into northern Mexico.

Smith Canyon

Smith Canyon is a tributary of Sonoita Creek located about three miles north of Patagonia, Arizona. It is within the Coronado National Forest, which is open to the public for recreational uses including hiking, camping, birdwatching, mountain biking, fishing and boating, and visiting historic areas (USFWS, 2021). Restoration treatments focus on the use of RDS to reduce erosion impacts and sediment pollution downstream and to assess effectiveness and impacts on ecosystem services. The area has been grazed in the past, but cattle are now excluded by fencing.

This watershed provides an opportunity for large-scale experimentation due to its structurally repetitive landscape consisting of roughly 90 physically similar sub-basins, each approximately 2–5 ha. The project includes treatment and control sites. The project is a collaboration between Borderlands Restoration Network, the Biophilia Foundation, and the US Geological Survey (USGS).

Babacomari Ranch

The Babacomari Ranch, located in Santa Cruz County, is one of the oldest land grant ranches in Arizona. The land is still used for cattle grazing, with management in collaboration with the University of Arizona and the National Resources Conservation Service. The Vaughn Canyon channel restoration includes “sandbagging,” the construction of 5 gabions, and other smaller RDS. Restoration has been a collaboration between Borderlands Restoration Network and the USGS, funded in part by the Walton Family Foundation (Petrakis et al., 2021).

Data

For this study we utilized a database of publications on El Coronado and Cuenca Los Ojos that had previously been compiled by the authors. The database was populated in early 2020 with publications provided by the property owners, supplemented by searches for property and parcel names (e.g., *El Coronado, Cuenca Los Ojos, Rancho San Bernardino*) in Scopus and Web of Science. Publications included peer-reviewed literature and student theses and reports. It has been maintained with additional contribution from the property owners and managers, who are aware of research conducted on the properties; publication alerts; and reference tracing. In 2021 we added sources related to Smith Canyon and Babacomari using the same methods. Eighteen records in the database were relevant to our research question and were included in this analysis.

Because not all impacts resulting from RDS-based restoration had been documented in the scientific literature, we supplemented the literature review with informal interviews with three key stakeholders who had installed, monitored, and/or observed the interventions for a period of at least 10 years and as much as 40 years (**Supplementary Table S1**,

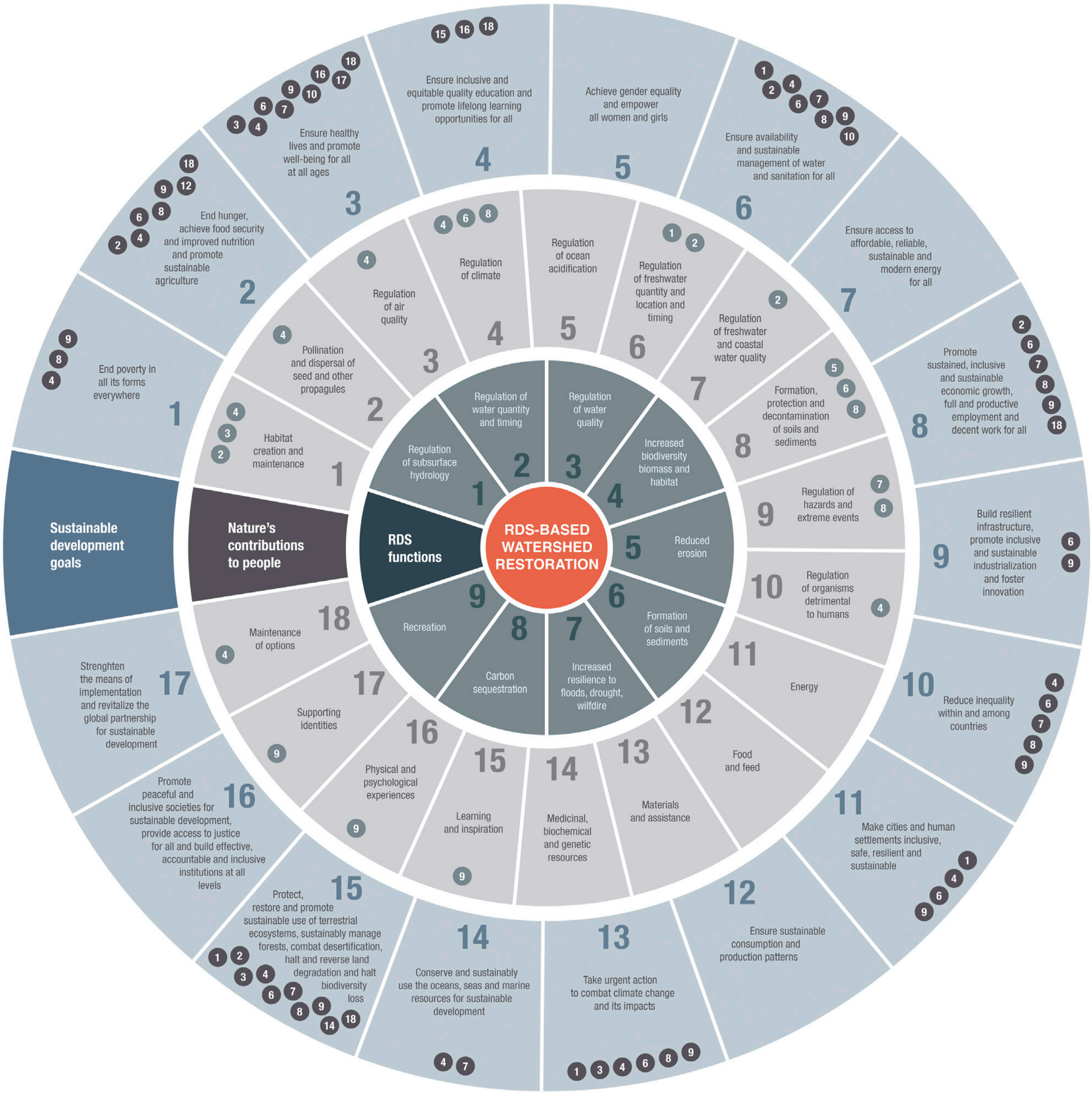

FIGURE 2 | Summary of the impact of RDS-based watershed restoration on ecosystem functions, on nature's contributions to people (NCPs), and on the United Nations Sustainable Development Goals (SDGs), showing the impact of each function on each NCP, and the contribution of the NCPs to each of the SDGs. Figure adapted from similar figure by Smith et al. (2019).

for interview data). The three interviewees were selected for their long history with and depth of knowledge about the case study sites. To provide structure and reduce bias, interviews were conducted using the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) categories of nature's contributions to people (NCP; see **Supplementary Table S2**) (Díaz et al., 2018) as a list of prompts. Interviewees were cued with each IPBES category and asked to provide their knowledge of relevant impacts, positive or negative. We note in our results which impacts were documented by published studies and which were observational.

Analytical Methods

Our objective for this study was to identify the multiple societal benefits of watershed-scale use of RDS as an NbS. Following

Smith et al. (2019), we analyzed our source material to identify the contribution of the intervention (RDS-based watershed restoration) in terms of ecosystem functions, nature-based solutions, and the Sustainable Development Goals. In this stage, we also evaluated the data for any indication of trade-offs between competing ecosystem services (Chausson et al., 2020).

In phase 1, we reviewed each publication that met the source criteria (see *Data*) and interviewee observations for evidence of ecosystem functions served by restored watersheds. We catalogued all relevant research findings in a spreadsheet, labeling the ecosystem function and noting the evidence to support each. We then consolidated the catalogued list of ecosystem services to standardize terminology and eliminate duplicates.

In phase 2, we utilized Díaz et al. (2018) framework for categorizing NCPs, a form of NbS, to group similar categories of the ecosystem services we identified in phase 1. Following methods developed by Smith et al. (2019), we associated each ecosystem function with the relevant NCP. These associations are indicated by numerical tags in **Figure 2**.

Finally, in phase 3, following Smith et al. (2019), we assessed the impacts of the RDS-based watershed restoration on the Sustainable Development Goals (SDGs). Each of the connections between the NCPs identified in phase 2 are indicated by numerical tags in **Figure 2**.

This study was conducted in adherence with the code of ethics of the Society for Ecological Restoration.

RESULTS

Ecosystem Functions

Here, we report the ecosystem functions served by RDS-based watershed restoration in drylands, as identified through our analysis of scientific literature and interviews associated with the four case study sites. Some ecosystem functions could be categorized under more than one NbS; for purposes of simplification, we have placed each according to the NbS with which it was most closely aligned (Díaz et al., 2018).

Formation, Protection, and Decontamination of Soils and Sediments

Erosion

Riparian areas and dry washes at our study sites were subject to significant erosion due to historical land uses. Incision of waterways ranged from a few centimeters to 9 m (Barry, 2014). The earliest RDS installed at El Coronado were constructed specifically for the purpose of controlling erosion (Barry, 2014). RDS help replicate the functions of healthy riparian ecosystems, where tree roots and vegetative debris would normally stabilize the soil, slow the speed of water, and reduce the energy of water moving downstream, reducing the water's ability to transport sediments. Some years after the installation of RDS, vegetation that has re-established can once again serve these functions (US Fish and Wildlife Service, US Bureau of Reclamation, US Forest Service, and Cross Watershed Network, 2018).

Modeling of the 7.7 km^2 Turkey Pen watershed at EC using the Soil and Water Assessment Tool (SWAT) found that 356–483 tons of sediment would likely be yielded from this watershed annually, given no management for erosion control, i.e., had no check dams been installed (Norman and Niraula, 2016). Modeling showed the installed RDS reduced sediment yield by 178–242 tons, or approximately by half. Conversely, a LiDAR study of the Babacomari Ranch found equal amounts of erosion (528 m^2) and deposition (497 m^2) surrounding a large gabion (~10 m wide); however, the authors note that this likely reflects the impacts of additional construction and rehabilitation of gabions during the study period, which may have increased erosion temporarily (Norman et al., 2019).

Formation of Soils and Sediments

In addition to preventing erosion, RDS can also help reverse the effects of past erosion by trapping sediments upstream of the structures (Fandel, 2016). This sedimentation causes streambeds to rise, reversing past incision of waterways (US Fish and Wildlife Service, US Bureau of Reclamation, US Forest Service. and Cross Watershed Network, 2018). Installation of RDS can be iterative and adaptive. If the structures reach their full capacity in trapping sediments, additional structures can be built atop older structures (Fandel, 2016), raising the streambed even further, ultimately (or ideally) reattaching the streambed to its floodplain.

Storing, Filtering, and Transforming Nutrients

Climate modeling has predicted increases in extent, frequency, and severity of wildfires in the western United States, leading to erosion and depletion of soils. Elsewhere, these fires have led to the permanent loss of N and C through geomorphic tipping points (Falk, 2013; Youberg et al., 2013). Delivery of nutrient-rich sediments to downstream water bodies negatively impacts water quality and aquatic ecosystems. A study of the ability of RDS to capture carbon and nutrients following wildfire found that both organic carbon and nitrogen in the sediments trapped post wildfire are twice (at 0.1-m depth) to 10 times (at 0.3-m depth) greater behind RDS than in off-channel soils in recently installed structures, though structures that had been in place for 30–40 years had in-channel organic carbon levels that more closely resembled off-channel levels (Callegary et al., 2021). RDS afforded the opportunity for quick reburial of mobilized biomass, soil organic matter, and charred organic matter following an occurrence of wildfire. If undisturbed, buried carbon is subject to soil and plant microbial processes, which result in some CO_2 being respired back into the atmosphere, some incorporated into plant roots and mycelia, and some mineralized or left inorganically in long term storage in soil, depending on conditions.

Regulation of Freshwater Quantity, Location, and Timing

Freshwater Quantity and Timing

Strong evidence documents positive impacts on freshwater quantity and availability throughout the year. Multiple studies have observed an overall increase in stream flow in areas where RDS have been installed throughout the landscape (Barry, 2014;

Buckley and Nabhan, 2016; Norman et al., 2016; Wilson and Norman, 2018). At El Coronado, flow volume increased 28% per unit of watershed area, relative to an untreated watershed (Norman et al., 2016). Duration of flow has also increased; a once-dry river now flows perennially for 9 km at CLO (Barry, 2014; Norman et al., 2019). These effects extend both downstream (5–10 km) and upstream (1 km) from the restoration area (Wilson and Norman, 2018). There have also been observations of pools of water present on hillsides during dry months (US Fish and Wildlife Service, US Bureau of Reclamation, US Forest Service. and Cross Watershed Network, 2018), and interviews indicate that the San Bernardino ciénega, historically a much larger wetland than in recent times, located in a watershed that has now been restored using RDS, has at least doubled in size since 2005.

Subsurface Hydrological Processes

Because RDS slow the flow of water over land and through waterways, they create conditions in which more water can soak into the soil, affecting subsurface hydrology. Using a heat transport method to measure infiltration flux, Fandel (2016) ran a series of weather simulations which, when combined with field measurements, found that a single gabion could increase total aquifer recharge, with simulations ranging from no impact to +225%, with the most likely scenarios depicting a 10.8% increase (Fandel, 2016). Modeling also shows increased subsurface hydraulic conductivity and accentuated lateral flow contributions to streamflow (Norman et al., 2019).

Observations indicate a decrease in the depth to water table due to sedimentation (unquantified) (Barry, 2014), which CLO land managers report occurred during a 15-years drought. Similarly, gabions installed in the upstream tributaries are likely impacting areas further downstream by raising the water table (Norman et al., 2014). Other observations include increased soil moisture on treated hillslopes, with pools of water observed during dry months (US Fish and Wildlife Service, US Bureau of Reclamation, US Forest Service. and Cross Watershed Network, 2018).

Regulation of Freshwater Quality

Freshwater Quality

One study anecdotally reported lower turbidity in a watershed treated with RDS than in an untreated watershed. During the study, a rainfall event of 5.2 cm occurred, with an average rate of 20.8 mm/h, over 150 times the average for the season. After the peak, observations showed no significant sand or silt deposits in the treated waterway, and the water flow was reportedly clear (Norman et al., 2016).

Regulation of Hazards and Extreme Events

Flood Resilience

In a model versus field measurement experiment, researchers found that the Soil Water Assessment Tool (SWAT) accurately (within 2.3%) predicted the hydrograph of a control watershed during several storm events of the seasonal monsoon in a southeast Arizona watershed. The SWAT model overestimated the hydrograph of the Turkey Pen watershed, which had been treated with over 2000 RDS, for which the SWAT model had not been calibrated, by 119.8%. Thus, stream gauges installed in the treated and untreated (control) waterways during monsoon season indicated that during inundation events the treated watershed had a lower runoff response and reduced peak flow, as modeled (Norman et al., 2016). Though baseflows are higher in treated streams, as discussed above (3.1.2.1), peak flows appear to be dampened. During the study, the aforementioned heavy rain event had little observable impact on the treated watershed. The authors note that after the event the only evidence of such heavy flooding was that the grass in the floodplain "had lain down" (Norman et al., 2016).

Drought Resilience

Relative to an untreated watershed, vegetation in a treated watershed at CLO was more resilient to dry conditions (Wilson and Norman, 2018). In the treated watershed, vegetation greenness, as measured by the Landsat Thematic Mapper using the normalized difference vegetation index (NDVI), was decoupled from spring precipitation, remaining greener than the control site even during seasons with low rainfall. The study found that RDS installation has increased water availability in the restored area, allowing vegetation to be less dependent on precipitation. Land managers also note that storing water in vegetated soil is preferable to storing water in open ponds because it is subject to less evaporation and therefore provides another adaptation to drought.

Catastrophic Wildfire

Because treated watersheds are greener and plant vegetation contains more water (Norman et al., 2019), it has been speculated that wildfires may be less severe in RDS-treated watersheds than in watersheds that are degraded and untreated. Land managers have speculated that the Horseshoe 2 fire in 2011, for example, which burned through the Turkey Pen watershed, may have been less intense due to restoration; however, other management treatments in and round the area make it difficult to assess drivers. Our research found no studies of the impact of RDS-based watershed restoration on the intensity of wildfires at the case study sites to date.

Habitat Creation and Maintenance

Biomass/Habitat

Studies of vegetative biomass and habitat have reported denser vegetation (US Fish and Wildlife Service, US Bureau of Reclamation, US Forest Service. and Cross Watershed Network, 2018), higher levels of greenness that increase over time, measured by the NDVI (Wilson and Norman, 2018); higher water content in vegetation, measured by the normalized difference infrared index (NDII) (Wilson and Norman, 2018); and increased perennial vegetation cover, assessed by field observations (Wilson and Norman, 2019). A time series of remote sensing data using the NVDI as a proxy for plant biomass showed that treatment sites increased in vegetation cover despite a general trend of below-average precipitation, in contrast to untreated sites, where vegetation decreased (**Figure 3**) (Norman et al., 2014).

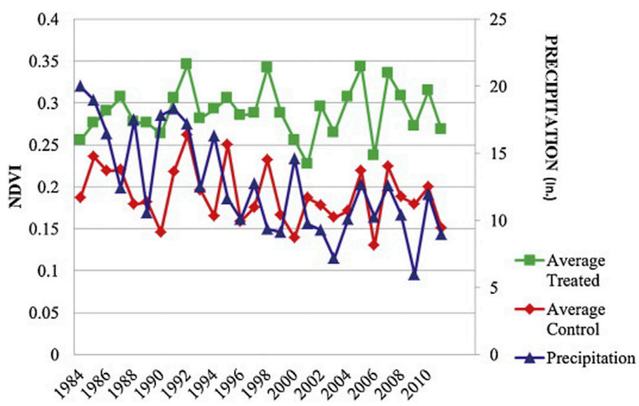

FIGURE 3 | Average NDVI plotted over time in San Bernardino (CLO) at treated and control sites, in relationship to annual precipitation (reprinted from Norman et al., 2014).

Biodiversity (Species, Habitats, and Genes)

In addition to an overall increase in biomass, there was also an increase in biodiversity (Cárdenas-García and Olguín-Villa, 2013; Barry, 2014). Using transects and vegetative surveying, researchers have observed an increased richness of plant species in riparian habitats, including more grass species (both native and non-native), more young age classes of trees that were not present before gabions were constructed, and many more aquatic plant species (Norman et al., 2014). Waterway restoration enabled the recolonization of native fish (Barry, 2014). Land managers at CLO also report increased presence of endangered species following watershed restoration, including jaguar (*Panthera onca*, Main, 2021), ocelot (*Leopardus pardalis*, pers. comm.), and black bear (*Ursus americanus*, Coronel-Arellano et al., 2018).

Regulation of Climate

Soil Carbon

Two studies have assessed the impact of RDS treatment on carbon sequestration in soil. A study at El Coronado assessed the ability of RDS of differing ages to capture soil carbon post-wildfire (Callegary et al., 2021). Results showed higher levels of soil carbon behind more recently constructed RDS: in-channel soil organic matter was twice (at 0.1-m depth) to 10 times (at 0.3-m depth) greater behind RDS than in off channel soils. At older structures, which had less remaining capacity to trap new post-wildfire sediments, there was little difference between in-channel and off-channel sediments. Mean additional carbon storage in the Turkey Pen watershed was 746–935 tons of CO_2 equivalent (tCO_2e) per ha of sedimentation behind RDS[1,2], an amount comparable to wetlands (Callegary et al., 2021).

TABLE 1 | Potential soil carbon storage as a result of RDS-based riparian restoration, reported in studies by Callegary et al. (2021) and Leger et al. (2009), converted to tCO_2e per ha

	Callegary et al. (2021)	Leger et al. (2009)
Low	746	75
Average		193
High	935	602

A separate study at the same site found 40% greater soil organic carbon in a treated channel compared to a control channel (Leger et al., 2019). Multiple soil samples in a 290 m^2 riparian area behind one RDS contained an average of 5.6 tCO_2e, but there was high variability (2.1-17.4 tCO_2e). The mean value equated to soil carbon storage of 193 tCO_2e per ha over an estimated 30 years of active sediment entrapment, a lower value than that found by Callegary et al. (2021) (**Table 1**). Differences in the studies' sample sizes and sampling methodologies and potential localized effects of post-fire, carbon-rich sedimentation confound the results and highlight the need for additional research.

Regulation of Micro-climate

Restoration of riparian vegetation that results from RDS installation creates a micro-climate along waterways, reducing local water and air temperatures through shading and evapotranspiration. A study of the endangered Yaqui catfish (*Ictalurus pricei*) found that the species was more likely to be present in areas with native riparian restoration and less likely where the waterways were bordered by cropland or shrubland (Hafen, 2018). Land managers have observed that the increase in native riparian vegetation following restoration creates cooler aquatic conditions and habitat more suitable to native species.

Physical and Psychological Experiences

Physical and Mental Health

A study that mapped social value preferences of residents of Patagonia, Arizona, within the Sonoita Creek watershed, which includes Smith Canyon, showed that across 12 measured values (aesthetics, biological diversity, cultural, economic, future generations, historical, intrinsic, learning, life sustaining, recreational, spiritual, and therapeutic), residents placed a high social value along Sonoita Creek and its main tributaries, particularly within the town of Patagonia and surrounding Patagonia Lake. Life sustaining, future generations, aesthetics, and biological diversity garnered the highest scores (Petrakis et al., 2020). Access to natural green spaces has been shown elsewhere to address both physical and mental health through multiple mechanisms (Staats, 2012).

Additionally, land managers have observed that volunteers and others who construct RDS find satisfaction in doing so. One restoration project manager who supervised a 10-person inmate crew wrote that "a unique rapport took shape, along with unanticipated levels of respect and pride in the work at hand" and detailed many anecdotes of comradery, humor, and realization of the positive impact that was being made (Seibert, 2015).

[1]To facilitate comparison, we have converted units of measurement from both studies to CO_2 equivalent (CO_2e) and hectares.

[2]Restored area was calculated to be 2.6 ha, a sum of sediment areas behind 2000 RDS in the 770-ha watershed (0.34% of land area).

Supporting Identities

Preserving and Utilizing Indigenous Knowledge

The RDS techniques deployed at the four study sites were adapted from techniques developed and utilized by Indigenous peoples in the region. The use of these techniques in modern conservation draws attention to the wisdom and skill of the peoples who inhabited these lands for centuries (Phillips, 2009). In addition, workshops offered by CLO to tribes in the region have enabled RDS techniques to be reintroduced on other tribal lands.

Preserving Ranching Heritage

For the last 300 years, the Sky Islands grasslands have been used by ranchers for sheep and cattle grazing (Cole and Cole, 2015). The use of unsustainable practices has led to land degradation and made the region less hospital for ranching, increasing the likelihood of conversion for other, even less sustainable uses, such as irrigated agriculture (Pool et al., 2014). Watershed restoration on ranchlands, coupled with sustainable ranching practices, can help make working lands more hospitable to wildlife and more viable over the long term, thereby maintaining ranching culture and heritage (Vásquez-León et al., 2003).

Maintenance of Options

Option Value

Protection of a wide variety of species, populations, and genotypes provides options for future generations to enjoy and utilize natural resources (Faith, 2016). Biodiversity also increases resilience to threats, such as a warmer climate (Isbell et al., 2015). Interviewees reported that current extractive economic activity (e.g., mining) is reducing options for the region's significant nature-based economy, which is based on tourism, outdoor recreation, sustainable harvest, and land restoration. By restoring landscapes in which nature-based economic activity can take place, RDS-based watershed restoration increases options for a sustainable economy, both now and in the future.

Contribution to Sustainable Development Goals

Following Smith et al. (2019), we traced the impacts of RDS-based watershed restoration and nature's contributions to people (Díaz et al., 2018) to the United Nations Sustainable Development Goals (SDGs). Relating ecosystem restoration to the SDGs underscores their potential contribution to factors that improve the lives of people. Results are shown in **Figure 2**.

Trade-Offs

A significant concern in NbS research is whether solutions to one problem incur trade-offs in the form of increased salience of another problem. For example, Chausson et al. (2020) found that in some cases NbS aiming for increased water availability came at a cost to protection against climate impacts. However, Chausson et al. (2020) suggested that interventions using natural or semi-natural ecosystems, as compared to non-native species, showed more synergies than trade-offs. Our findings support this conclusion. The only trade-off we identified was temporal: one study observed a construction-related decrease in vegetation near gabions until approximately 2–4 years after installation, after which there was a net increase in vegetation (Wilson and Norman, 2018). We found no evidence of consequential trade-offs among these factors in publications or interviews with stakeholders. On the four case study sites, we found water availability, biomass cover, erosion control, flood resilience, and carbon sequestration were mutually increased as a result of the intervention.

Costs

A first approximation of costs can be calculated using information from an RDS construction project conducted at the Babacomari site. Figures provided by the Borderlands Restoration Network indicate that medium-sized check dams could be built for approximately $65 USD each in 2019, assuming a work crew (1 foreman, 5 laborers) dedicating 2.25 h per medium-sized check dam (D. Seibert, pers. comm.). Given the density of structures installed at EC and CLO, labor costs for a 1-km stretch of waterway could be restored for approximately $1,600 USD. These calculations do not include the costs of any materials or equipment. Consistent with many conservation interventions (Cook et al., 2017; Iacona et al., 2018), further documentation of costs is necessary to fully assess the costs and benefits of restoration.

DISCUSSION

Summary

RDS have been utilized around the world, for centuries or longer (Abbasi et al., 2019; Norman, 2020). Their use to plug gullies, control erosion, increase sedimentation, trap precipitation, increase water infiltration, and make land more productive and suitable for agriculture has been documented in various manuals and solutions databases, such as the World Overview of Conservation Approaches and Technologies (WOCAT) Sustainable Land Management database (WOCAT, 2021). Yet there has been little attention on their potential use as an NbS for climate mitigation and adaption.

Over 40% of the earth's land area is classified as desert, semi desert, arid and semi-arid grasslands or rangelands, containing 44% of the world's cultivated systems (Reid et al., 2005). One third of the earth's human population inhabit these areas, 40% of whom have livelihoods directly affected by desertification. Ephemeral and intermittent streams, the lifeblood for these areas, make up more than half the combined length of all rivers and streams globally (Acuna et al., 2014). The evidence from our case studies suggests that RDS-based interventions offer a means to adapt to the conditions of an altered climate, including higher temperatures, more variable precipitation, and more extreme weather events. More research is necessary to assess the range of eligible application, but their use in drylands across the globe suggest they could provide relief and resilience for some of the 2.1 billion human inhabitants of drylands worldwide.

Addressing Barriers to Replication and Scalability

This Frontiers special research topic, Nature-Based Solutions for Natural Hazards and Climate Change, emphasizes the need for research to identify and scale replicable options for NbS. As conservation practitioners and funders, we are particularly interested in issues of replicability and scalability and ask: if RDS-based watershed restoration is feasible, cost-effective, and impactful, why is it not more widely utilized?

Currently, there is no analysis of barriers to restoration in the Sky Islands region, so we used the United Nations Decade on Ecosystem Restoration (hereafter, UN Decade) strategy document (United Nations Environment Program, 2020), which identifies global barriers to widespread ecosystem restoration, as a guiding framework. Here we elaborate on five of the barriers: 1) limited awareness of land degradation and the benefits of restoration, 2) lack of legislation, policies, and regulation, 3) limited technical capacity, 4) limited finance, and 5) need for more research.[3]

Limited Awareness of Land Degradation and the Benefits of Restoration

Because land degradation in the Sky Islands has occurred over a period of 200–300 years, changes to the land can be less obvious within a single generation (i.e., shifting baseline, Pauly, 1995). As a result, there is limited awareness of the historical functionality of ecosystems, the extent and costs of degradation, or the potential societal benefits that could accrue with restoration (United Nations Environment Program, 2020).

Positive impacts of RDS-based watershed restoration have been documented in scientific papers across multiple disciplines. Our aim with this paper was to catalog and synthesize the evidence in support of the intervention, based on work at four case study sites that have collectively been the subject of research in multiple disciplines. We found strong evidence for impacts on freshwater quantity and timing. Highlights include a 28% increase in watershed flow volume (Norman et al., 2016) and longer annual duration of flow (Barry, 2014). We also found positive impacts on formation of soils and sediments, freshwater quantity and timing, resilience to flooding and drought, vegetation, biodiversity, pollination and seed dispersal, and human use of the landscape. We hope this paper and associated outreach materials help address the barrier of awareness of the societal value and benefits that accrue as a result of RDS-based watershed restoration.

Lack of Legislation, Policies, and Regulation

The UN Decade identifies a lack of institutional mechanisms that incentivize investments in large-scale restoration (United Nations Environment Program, 2020). The IUCN Global Standard for Nature-based Solutions (IUCN, 2020) provides definitional criteria, a systematic learning framework, and recommendations for governance. In addition, the guiding principles offered by the Nature-Based Solutions Initiative, which have been endorsed by 20 organizations (Nature Based Solutions Initiative, 2020) and which are described below, can be used to inform the development of new legislative, policy, and regulatory tools in the region.

1) *NbS are not a substitute for a rapid fossil fuel phase-out.* Carbon markets are the most likely source of short-term funding for NbS (Seddon et al., 2021). However, over the medium to long term, we recommend constraint of restoration in the Sky Islands as an offset for carbon emissions elsewhere, with emphasis instead on phasing out fossil fuels.
2) *NbS involve the protection and/or restoration of a wide range of naturally occurring ecosystems.* Recent global NbS implementation has included afforestation at the expense of other vital ecosystems, such as grasslands. Restoration of woody vegetation associated with RDS installation in the Sky Islands should be done in riparian and wetland ecosystems where trees grew in the past or where they regenerate naturally.
3) *NbS are implemented with full engagement and consent of Indigenous peoples and local communities, apply social safeguards, and build human capacity to adapt to climate change.* RDS originated in the region as Indigenous technology, and their use resonates with many local communities. IUCN guidance provides best practices for Indigenous and community conservation areas (Borrini et al., 2013) and privately protected areas (Mitchell et al., 2018).
4) *NbS sustain, enhance, or support biodiversity.* Project goals at case study sites included restoration of habitat for threatened and endangered wildlife, providing a template for regional efforts. RDS can support different constellations of goals and resources, but attention must be given to potential trade-offs between biodiversity and ecosystem services.

Limited Technical Capacity

Limited technical capacity to design and implement restoration initiatives can hinder their widespread use (United Nations Environment Program, 2020). As an example of low-tech, process-based restoration (Wheaton et al., 2019), many of the structural interventions of RDS-based watershed restoration can be built by hand. Their placement requires expertise – though not to the same degree as engineered solutions – but RDS can be built by volunteers or workers with few pre-existing labor market skills. Much of the wisdom on placement and construction of RDS originated in Indigenous communities, where knowledge and skills were communicated orally and experientially. Organizations such as Cuenca Los Ojos and the Borderlands Restoration Network are using similar methods, through workshops, peer-to-peer exchanges, volunteer workdays, and youth trainings, to share the knowledge today.

While low-tech methods are not a complete solution to watershed restoration, their use opens opportunities for community engagement in restoration, which has potentially transformational effects on communities and their local political economy (Pritzlaff, 2018) and the potential for restoration across a larger land area than would otherwise be possible. As an illustration, low-tech, process-based restoration

[3]The UN Decade identifies an additional barrier that we do not discuss here: low pressure to invest in ecosystem restoration relative to other sectors, such as health care or education (United Nations Environment Program, 2020). We recognize the extent of this issue in the US, where environmental causes received only 12% of philanthropy, as of 2009 (Ramutsindela et al., 2011), but lack sufficient knowledge of other sectors in the Sky Islands to compare.

techniques are considered applicable in wadable streams (i.e., fifth order streams or less), which account for roughly 90% of the perennial stream length in a typical drainage network (Wheaton et al., 2019, p.4).

Limited Finance

Lack of financial resources for conservation is a perpetual concern (Clark, 2007), and investment can be difficult to find due to mismatches in terms of timescale and beneficiary. Restoration at our four case study sites has been funded primarily through philanthropy, including funding from private landowners, charitable foundations, and government agencies. Funding from these sources are limited, however, and additional sources of revenue are needed. One option is payments for ecosystem services (PES) schemes, including carbon credits. Based on more than 2 years of investigation into this topic, including consultation with researchers, restoration professionals, and experts on market creation, we believe carbon markets are a viable financing mechanism for three reasons.

1) Preliminary evidence suggests carbon can be sequestered by RDS. Based on our case study sites, anywhere from 70-930 tCO_2e per hectare can be stored in soils alone, which at the upper end is comparable with sequestration rates in wetlands, which currently receive substantial investment from carbon markets (Zomer et al., 2017)
2) Some suitable carbon markets are already in place, and new markets are coming online each year. Carbon markets are seeing record market volume and value in 2021, with markets on track to reach $1 billion in transactions this year (Ecosystem Marketplace, 2021).
3) Carbon market funding is scalable. We estimate that 37 million hectares of watersheds in the Sky Island region need restoration, potentially storing 13.8 million tCO_2e. Callegary et. al. (2021) estimated 40 million tCO_2e could potentially be stored in restored watersheds of forested regions of the entire southwestern United States. These figures approximate carbon captured in soils, not carbon stored in standing trees of restored riparian areas, where some researchers suggest most carbon is stored (please see **Supplementary Material** (Calculations)). Carbon sequestration is but one potential benefit of RDS-based watershed restoration, but with demand for carbon credits growing over time, carbon markets offer a feasible mechanism to finance regional-scale restoration.

Need for More Research

There is evidence of positive impacts of RDS-based watershed restoration on hazard mitigation and climate resilience at our case study sites. Yet many research gaps remain. At the case study sites, there has been less attention on impacts on freshwater quality, biodiversity, and resilience to catastrophic wildfire. Given anticipated impacts on these factors due to climate change, these gaps urgently need to be filled. Other ecosystem functions, such as pollination, seed dispersal, and air quality, have not yet been investigated.

More research to assess the carbon sequestration potential of RDS-based watershed restoration is necessary, particularly to understand the contribution of both above ground and below ground plant biomass and fallen large wood. Additionally, most studies on the ecosystem impacts of restoration have been retroactive. More experiments like the study recently initiated at Smith Canyon are needed (Petrakis et al., 2021).

Finally, although we reported some information about the cost of constructing RDS, we lack sufficient documentation of the costs of watershed-scale restoration, information that is necessary to assess the financial viability of PES programs and carbon market-financed restoration projects.

CONCLUSION

Dryland watershed restoration provides a range of environmental and social benefits. This low tech restoration method has been practiced for centuries with success, and the skills can easily be taught to the inhabitants of the areas where it is most needed, providing local employment and livelihood opportunities. The materials needed for construction are often found on site. The practices themselves are adaptable, should field monitoring suggest alterations based on changing hydrologic or other conditions.

Dryland watershed restoration was not included by Fargione et al. (2018) as a natural climate solution for the United States. We believe the results of the case studies presented here illustrate the range of ecosystem functions served by RDS-based restoration and provide a basis for exploring opportunities to pursue dryland watershed restoration as nature-based solution to climate mitigation and adaptation.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article and its **Supplementary Material**, and further inquiries can be directed to the corresponding author.

ETHICS STATEMENT

Ethical review and approval was not required for the study on human participants in accordance with the local legislation and institutional requirements. Written informed consent for participation was not required for this study in accordance with the national legislation and the institutional requirements.

AUTHOR CONTRIBUTIONS

Both authors contributed to review and summary of published literature and to writing the manuscript. JG developed the theoretical framework, conducted interviews, and analyzed interview data. RP analyzed carbon sequestration and carbon finance potential. Both authors have made substantial, direct, and intellectual contribution to the work and approved it for publication.

ACKNOWLEDGMENTS

The authors would like to thank the proprietors of the case study sites for their support of the research cited herein and their ongoing work restoring the Sky Islands ecosystem.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2021.679189/full#supplementary-material

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Conflict of Interest: RP is founder and chairman and JG is president/CEO of the Biophilia Foundation, which has funded some of the work conducted at the case study sites. JG has worked as a conservation strategist for a coalition of organizations in the region that includes Cuenca Los Ojos, the Biophilia Foundation, and the Borderlands Restoration Network, which are closely involved with the case study sites.

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frontiers | Frontiers in Environmental Science

GENERAL COMMENTARY
published: 01 April 2022
doi: 10.3389/fenvs.2022.853684

Commentary: Dryland Watershed Restoration With Rock Detention Structures: A Nature-Based Solution to Mitigate Drought, Erosion, Flooding, and Atmospheric Carbon

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Keywords: ecosystem services, natural climate solution, erosion control structures, riparian, restoration, climate adaptation and mitigation, resilience

A Corrigendum on

Dryland Watershed Restoration with Rock Detention Structures: A Nature-based Solution to Mitigate Drought, Erosion, Flooding, and Atmospheric Carbon
by Gooden J., and Pritzlaff R. (2021). Front. Environ. Sci. 9:679189. doi:10.3389/fenvs.2021.679189

OPEN ACCESS

Edited by:
Borja Gonzalez Reguero, University of California, Santa Cruz, United States

Reviewed by:
George Nick Zaimes, International Hellenic University, Greece
Thomas Sisk, Northern Arizona University, United States

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Specialty section:
This article was submitted to Interdisciplinary Climate Studies, a section of the journal Frontiers in Environmental Science

Received: *12 January 2022*
Accepted: *25 February 2022*
Published: *01 April 2022*

Citation:
Norman LM (2022) Commentary: Dryland Watershed Restoration With Rock Detention Structures: A Nature-Based Solution to Mitigate Drought, Erosion, Flooding, and Atmospheric Carbon. Front. Environ. Sci. 10:853684. doi: 10.3389/fenvs.2022.853684

INTRODUCTION

In their paper, the authors describe studies of rock detention structures (RDS) at four properties and create an argument to acknowledge RDS as nature-based solutions (NbS) based on their documented climate mitigation and adaption potential (Gooden and Pritzlaff, 2021). The adoption of national and international strategies to mitigate climate change and accompanying desertification and drought is dependent on decision-makers understanding the significance of impacts, acting on available anticipatory science, and assuring a commitment to financial investments (Bradford et al., 2018). As such in this invited commentary, I provide additional research describing RDS scalability throughout landscapes, perseverance over time, and contributions to a restoration stewardship economy that supports RDS as NbS.

Contemporary studies conducted on RDS span the United States-Mexico border in North America, describe multiple benefits they provide for people (ecosystem services), and are globally applicable to arid land environments (Norman, 2020). The installation of RDS sets biophysical cycles into motion that persist through centuries, providing anticipatory and restorative functions (Norman, 2020). Additionally, RDS allow capacity-building and risk mitigation that can alleviate climate change impacts in socio-environmentally vulnerable regions (Norman et al., 2021a; Norman et al. 2021a; Norman et al., 2021b). RDS provide sustainable NbS to address climate change and can be implemented to protect, sustainably manage, and restore ecosystems (International Union for Conservation of Nature, 2021).

SCALABILITY IN SPACE AND TIME

Over the past 1,000+ years, human populations in the arid North American Southwest have left archeological evidence of RDS across the landscape, but long-term ecohydrological, demographic and socio-economic drivers and impacts have yet to be explored in an integrative and iterative way (J. Dean, University of Arizona, written communication 11/12/2021; Fish and Fish, 1984; Hall et al., 2013; Norman, 2020). The scaling of RDS-NbS interventions and benefits from local to watershed scales and throughout time requires objective science. Researchers are working with land managers and restoration practitioners to establish experiments at multiple RDS installations throughout the Madrean Archipelago Ecoregion of North America to quantify the effects of RDS-NbS on hydrology, ecology, and soil productivity (Fandel et al., 2016; Norman et al., 2016; Norman et al., 2017; Norman et al., 2019; Norman, 2020; Callegary et al., 2021; Coy et al., 2021; Norman et al., 2021a; Wilson et al., 2021; Freimund et al., 2022).

Quantitative models and remotely sensed imagery expand site-based experiments at RDS into larger regional watersheds and allow forecasting and back-casting to extrapolate over time (Norman L. M. et al., 2010; Norman et al., 2014; Norman and Niraula, 2016; Norman et al., 2017; Wilson and Norman, 2018; Norman et al., 2019; Norman, 2020; Norman, 2021; Norman et al., 2021b; Petrakis et al., 2021; Lara-Valencia et al., 2022). RDS support vital ES that address societal problems, including flood regulation; water regulation, purification, and provisioning; habitat provisioning; erosion regulation, carbon sequestration and storage; social value and climate regulation (Norman, 2020; Norman et al., 2021b). These ES protect biodiversity and mitigate climate change with cumulative and multiplicative feedback effects that sustain their positive impacts in the long term (Norman L. et al., 2010; Norman et al., 2014; Norman et al., 2016; Norman and Niraula, 2016; Norman et al., 2017; Norman et al., 2019; Norman 2020; Norman et al., 2021a; Norman et al., 2021b).

WATER AND CLIMATE RESILIENCE INFRASTRUCTURE

The focal paper (Gooden and Pritzlaff, 2021) translates the ecosystem services of rock detention structures (RDS; Norman, 2020) into Nature-based Solutions (NbS) and relates them to sustainable development goals. NbS is a term that is often used to re-frame ecosystem services and green infrastructure projects, to make them more politically palatable (O'Sullivan et al., 2020). Like green infrastructure, RDS rely on vegetation, soils, and natural processes to manage water and create healthier environments. However, unlike green infrastructure, which promotes passive rainwater harvesting to retain water in the built environment, RDS are established to detain water (not retain it) in more remote areas; allowing water to slowly pass through, infiltrate the soils and regenerate landscapes (Norman et al., 2014; Norman et al., 2016; Norman and Niraula, 2016; Norman et al., 2017; Wilson and Norman, 2018; Norman et al., 2019; Norman 2020; Norman et al., 2021a; Norman et al., 2021a; Norman et al., 2021b).

EXAMPLE: COSTS AND BENEFITS

Costs: Check Dams

One of the disadvantages to using RDS is the associated costs and investment needed to secure them. It takes an average of 16 h of labor to install a 1-m-high RDS (check dam; **Figure 1**; J. T. Austin, former owner El Coronado Ranch, oral communication, 12/17/2021; Norman et al., 2022). Approximately 2000 RDS are installed on a remote, 769-ha forested watershed in southeastern Arizona (Norman and Niraula, 2016). Including current wages, equipment, and associated estimates, it costs approximately ~$2,210 per ha to treat a watershed based on this established rate (~2.6 check dams/ha; Callegary et al., 2021).

In this example, I use back-of-the-envelope calculations to roughly extrapolate how much it would cost to preemptively treat all Federal and Tribal riparian areas in the state of Arizona (33,182 ha) with check dams. This ballpark figure entails installing 86,273 check dams with an estimated cost of $73M. For comparison, climate-related disasters in Arizona spurred Legislation of $100M for recovery and support efforts to help deal with damages related to post-fire flooding in 2021 (Office of the Governor, 2021).

Benefits: Ecosystem Services, Economics, and Equity

There are a lot of advantages to be realized from using RDS. Thomas et al. (2016) estimated the economic impacts of restoration associated with Federal lands, to be between 13–32 job-years and $2.2-$3.4M for every $1M spent.

In the example provided above for costs and using economic impacts of restoration documented by Thomas et al. (2016), I estimate the potential return on this ecological restoration investment scenario of installing check dams in Federal riparian areas ($73M) to produce the equivalent of >1,000 job-years and >$160M of economic output to local, regional, and national economies. And, given the durability of RDS, with some ongoing investment in repair and maintenance, the initial investment will provide long-term benefits (Norman L. M. et al., 2010; Norman 2020; Norman et al., 2021a; Norman et al., 2021b).

Based on targeted science research and results of the Aridland Water Harvesting Study (Norman, 2020; Norman et al., 2021a; Norman et al., 2021b), this example scenario (to install ~86,273 check dams in Federal riparian areas of Arizona) could also:

✓ sequester ~7.5 M tons [0.0075 Pg (7.5 Tg)] of atmospheric C in the soil storage (Norman and Niraula, 2016; Callegary et al., 2021);
✓ maintain or increase vegetation and biomass—with extended growing seasons and using stored soil moisture (Norman et al., 2014; Wilson and Norman, 2018; Wilson et al., 2021), further increasing C sequestration;
✓ extend ephemeral duration and surface-water availability (Norman et al., 2016; Norman and Niraula, 2016; Norman et al., 2017);

FIGURE 1 | Author sitting on a 30-year-old check dam installed at the El Coronado Ranch study area, Chiricahua Mountains, southeastern Arizona, United States (photo by Gerry Norman, Oct. 2021).

✓ mitigate floods and associated emergency response expenditures (Norman L. M. et al., 2010; Norman L. et al., 2010; Norman and Niraula, 2016; Norman et al., 2021b; Freimund et al., 2022);
✓ promote lateral flows and onsite storage of water (Fandel et al., 2016; Fandel, 2016; Norman et al., 2016; Norman et al., 2019);
✓ control erosion and nonpoint source pollution, improving water quality (Norman L. M. et al., 2010; Norman L. et al., 2010; Norman and Niraula, 2016; Norman et al., 2017; Norman et al., 2019); and
✓ reduce ambient temperatures (Norman et al., 2021).

Water-related ES in drylands also increases the quality of life for socio-environmentally vulnerable communities (Norman L. et al., 2010, Norman et al., 2012; Norman et al., 2013; Villarreal et al., 2013). A restoration economy, based on improved hydrology in degraded waterways and associated riparian areas, can also be created that improve lives and livelihoods in restored areas (Adams, 2016; Norman et al., 2021a; Norman et al., 2022).

DISCUSSION

Over the past decade, a multi-disciplinary landscape-scale study has quantified anticipatory and restorative watershed functions of Rock Detention Structures (RDS) installed in the Madrean Archipelago Ecoregion of North America (Norman, 2020). In this commentary, I reference these larger temporal and spatial study extents to underpin RDS interventions as Nature-based Solutions (NbS) as presented in Gooden and Pritzlaff (2021), that restore dryland channels with impartial shares of social, environmental, and economic benefits.

Our planet needs solutions to mitigate impacts from our rapidly changing climate, and the ecosystem services of RDS justify the inclusion as NbS, useful to increase carbon storage and sequestration, increase water quality and quantity, buffer flood events, improve vegetation health and biodiversity, and help address global warming (Norman 2020; Norman et al., 2021a; Norman et al., 2021b). The focal paper proposes RDS as a feasible, cost-effective NbS that can contribute to climate mitigation (Gooden and Pritzlaff, 2021).

RDS-NbS are already being used to enhance preparedness, response, and resilience in vulnerable communities along the United States-Mexico border, where a restoration stewardship economy fosters hope, nourishes livelihoods, and establishes valuable ecosystem services via grassroots efforts (Norman et al., 2021a). Investments in RDS-NbS avert risk, damage, and cost associated with drought and flooding (Norman L. et al., 2010; Norman et al., 2014; Norman et al., 2016; Norman et al., 2019; Norman et al., 2021b). Moreover, capital appreciation, described herein, can double or triple restoration investments on Federal lands (Thomas et al., 2016), RDS-NbS provide a huge suite of benefits, and the structures can persist for millennium (Norman 2020).

AUTHOR CONTRIBUTIONS

The author confirms being the sole contributor of this work and has approved it for publication.

FUNDING

Funding for this commentary was provided by the Land Change Science Program of the U.S. Geological Survey. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

ACKNOWLEDGMENTS

The author thanks all the partners in the Sky Island Restoration Collaborative and appreciates reviews by Julio Betancourt and Michele Girard.

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Conflict of Interest: The author declares that she was an adviser of and has co-authored with Dr. Pritzlaff, who also contributed funding to some of the restoration work studied herein.

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frontiers in Earth Science

ORIGINAL RESEARCH
published: 04 November 2021
doi: 10.3389/feart.2021.676059

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The 'Rocket Framework': A Novel Framework to Define Key Performance Indicators for Nature-based Solutions Against Shallow Landslides and Erosion

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OPEN ACCESS

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Specialty section:
This article was submitted to Geohazards and Georisks, a section of the journal Frontiers in Earth Science

Received: *05 March 2021*
Accepted: *14 October 2021*
Published: *04 November 2021*

Citation:
Gonzalez-Ollauri A, Munro K, Mickovski SB, Thomson CS and Emmanuel R (2021) The 'Rocket Framework': A Novel Framework to Define Key Performance Indicators for Nature-based Solutions Against Shallow Landslides and Erosion. Front. Earth Sci. 9:676059. doi: 10.3389/feart.2021.676059

The idea of nature providing solutions to societal challenges is relatively easy to understand by the layperson. Nature-based solutions (NBS) against landslides and erosion mostly comprise plant-based interventions in which the reinforcement of slopes provided by vegetation plays a crucial role in natural hazard prevention and mitigation, and in the provision of multiple socio-ecological benefits. However, the full potential of NBS against landslides and erosion is not realised yet because a strong evidence base on their multi-functional performance is lacking, hindering the operational rigour of NBS practice and science. This knowledge gap can be addressed through the definition of repositories of key performance indicators (KPIs) and metrics, which should stem from holistic frameworks facilitating the multi-functional assessment of NBS. Herein, we propose the 'rocket framework' to promote the uptake of NBS against landslides and erosion through the provision of a comprehensive set of indicators which, through their appropriate selection and measurement, can contribute to build a robust evidence base on NBS performance. The 'rocket framework' is holistic, reproducible, dynamic, versatile, and flexible in helping define metrics for NBS actions against landslides and erosion along the NBS project timeline. The framework, resultant from an iterative research approach applied in a real-world environment, follows a hierarchical approach to deal with multiple scales and environmental contexts, and to integrate environmental, eco-engineering, and socio-ecological domains, thus establishing a balance between monitoring the engineering performance of NBS actions against landslides and erosion, and the wider provision of ecosystem functions and services. Using a case study, and following the principles of credibility, salience, legitimacy, and feasibility, we illustrate herein how the 'rocket framework' can be effectively employed to define a repository with over 40 performance indicators for monitoring NBS against landslides and erosion, and with over 60 metrics for establishing the context and baseline upon which the NBS are built and encourage their reproduction and upscaling.

Keywords: green infrastructure (GI), soil bio-engineering, hydro-meteorological hazards, open-air laboratories (OALs), ecosystem services, monitoring, indicators and metrics

INTRODUCTION

Nature-based Solutions (NBS) can be defined as "actions to protect, sustainably manage and restore natural and modified ecosystems that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits" (Cohen-Shacham et al., 2016). They are an emerging concept referring to actions and interventions that use natural features and processes to address environmental problems at different scales. The uptake of NBS is rapidly increasing, in part because the idea of nature providing solutions is simple to understand by the layperson (Cohen-Sacham et al., 2019). However, there still are several challenges to overcome prior to realising the full potential of NBS (Nelson et al., 2020). These challenges stem from the lack of operational rigour in NBS practice and science (i.e., weak knowledge of NBS design, implementation, and evaluation), and from the lack of homogeneity between NBS concepts and frameworks, which have been generally established around eclectic disciplines such as urban sustainability, ecosystem-based and climate adaptation approaches, or conservation ecology (e.g., Raymond et al., 2017; Ruangpan et al., 2020). Still, the uncertainty associated with NBS performance is acknowledged as the greatest limitation to overcome by the existing NBS frameworks, placing monitoring activities at their core to promote the generation of a robust evidence base (Raymond et al., 2017; Woroniecki et al., 2019). Monitoring should be a transversal process across the NBS project stages, but it should also be strategically devised as a platform to gather evidence on NBS performance once specific actions have been deployed. This evidence base should convey information and confidence across the public and private sectors upon NBS performance in complex, dynamic systems, and it should consolidate the standardisation and upscaling of NBS practice, including specifications and benchmarks (e.g., Kabisch et al., 2016; Angelakoglou et al., 2019; Nelson et al., 2020).

Several frameworks have been proposed over the last few years to assess NBS projects, to define alternative courses of action, and to address the lack of evidence on NBS performance (for review, see Narayan et al., 2016, Wendling et al., 2018 or Shah et al., 2020). These frameworks agree that NBS can provide both socio-ecological and environmental benefits (and co-benefits) to a wide range of stakeholders, when deployed effectively in a given context. NBS performance has been widely proposed to be assessed using indicators from the economic, social, and environmental domains (e.g., Kabisch et al., 2016). However, most frameworks have emphasised the social domain over the ecological/environmental domain (Shah et al., 2020). This is likely because most research-oriented NBS projects have been focusing on climate adaptation and resilience within urban environments (e.g., UNaLab, Eklipse, RECONECT). Also, the change in paradigm experienced by the field of nature conservation in the late 2000s, which evolved from focusing solely on nature to focus on people and nature, has taken NBS beyond the traditional conservation and management principles by re-focusing the debate on humans (Eggermont et al., 2015; Cohen-Sacham et al., 2019). However, NBS are not just strongly connected to socio-ecological ideas such as urban sustainability, climate adaptation, or ecosystem-based management approaches, the concepts of green infrastructure and ecological engineering may, in fact, be closest to NBS (Eggermont et al., 2015; Fernandes and Guiomar, 2018); especially when NBS are sought to manage hydro-meteorological hazards (HMHs), such as landslides and erosion, which require the intervention and modification of the hazard-prone ecosystems to address these societal challenges (e.g., Schiechtl and Stern, 1996; Morgan, 2004).

Both green infrastructure (GI) and ecological engineering strive to merge engineering principles and ecological knowledge to protect, restore, modify and/or build new ecological systems that provide services that would otherwise be provided through more conventional, 'grey' or 'hybrid' engineering (Mitsch, 2012; Anderson and Renaud, 2021). Although NBS actions belonging to the typologies of GI and ecological engineering comprise intrusive design and management of new or existing ecosystems, they can also maximise the delivery of key functions and services to human communities (e.g., Costanza et al., 1996). In fact, the consideration of ecological engineering (Eco-engineering) concepts and principles generates a unique opportunity to introduce natural dynamics into the conceptualisation of infrastructure, which is one of the key features of NBS. These aspects have been originally explored by researchers working on NBS for water treatment and for the resilience of coastal ecosystems against storm surges, flooding and erosion (e.g., Pontee et al., 2016; Thorslund et al., 2017; Reguero et al., 2018). As NBS actions based on eco-engineering follow engineering principles (i.e., ideas, rules, or concepts to be kept in mind when solving engineering problems), this opens up an exciting opportunity to utilise existing performance indicators framed in well-established and standardised engineering monitoring protocols (e.g., the Eurocodes but also e.g., Garcia-Rodriguez et al., 2019). To our knowledge, the eco-engineering performance of NBS actions against landslides and erosion has not been explicitly considered within NBS frameworks yet. Although the performance metrics for HMHs are usually strong, either in societal terms or environmental aspects (Maes et al., 2016; Woroniecki et al., 2019), they typically lack holistic perspective without including the eco-engineering domain.

Many societal challenges associated with environmental problems, and which NBS strive to address, have an engineering component and can thus be regarded as engineering problems occurring within an ecological context. The construction of wetlands to treat water at the catchment level is one example of how the combination of hydraulic engineering, water science, and plant ecology can effectively address a socio-ecological challenge such as eutrophication (e.g., Costanza et al., 1996; Thorslund et al., 2017). Wetlands and reefs can also be engineered to protect coastal ecosystems against flooding and coastal erosion (e.g., Pontee et al., 2016; Reguero et al., 2018). HMHs, such as landslides and erosion, are another example of a societal challenge that can be managed with NBS that merge civil engineering and ecological principles and knowledge (e.g., Stokes et al., 2014). Shallow landslides and erosion produce dramatic episodes of soil mass wasting and severely damage human life and property in sloping areas worldwide. Both phenomena are mostly triggered by rainfall, which will be more frequent and intense due to climate change (Sidle and Bogaard, 2016). Yet, their impact and severity can be reduced with NBS comprising plant-based interventions in which the mechanical and hydrological reinforcement of slopes provided by vegetation plays a crucial role in the prevention, mitigation, and management of the hazards (Gonzalez-Ollauri and Mickovski, 2017a; Gonzalez-Ollauri and Mickovski, 2017b; Gonzalez-Ollauri and Mickovski, 2017c; Gonzalez-Ollauri and Mickovski, 2017d). Indicators that would confirm the effectiveness of vegetation in combatting landslides and erosion would include concepts from geotechnical engineering such as soil strength, permeability, erodibility and similar, but also concepts from environmental sciences such as land coverage or species richness. Each indicator would be a measure (metric) based on verifiable data that condenses complexity and conveys information (Haase et al., 2014). Plant-based NBS against landslides and erosion may also provide multiple socio-ecological benefits, but these will not be delivered unless the envisaged eco-engineering performance of the NBS is met. The latter stresses that the interconnectivity of context (e.g., habitat, ecosystem, landscape, etc.) in which NBS actions against landslides and erosion are deployed makes the adequate provision of eco-engineering functions essential to promote the provision of economic, social, and environmental functions and co-benefits. However, this has not been thoroughly explored in the context of NBS managing HMHs, in general, and landslides and erosion, in particular, due to the lack of holistic frameworks that foster monitoring activities through the identification of multi-functional KPIs helping to build a robust evidence base on NBS performance (Cohen-Sacham et al., 2019).

The aim of this paper is to propose a novel, holistic, and reproducible framework to define metrics and key performance indicators for monitoring NBS actions against landslides and erosion along their project timeline. The NBS project timeline incorporates all the steps undertaken pre- and post-NBS implementation to establish the project objectives, understand local conditions, conceptual and detailed design of the NBS, and choose the appropriate assessment approaches for performance, sustainability, and cost-effectiveness. The proposed framework, resultant from an iterative research approach applied in a real-world environment prone to landslides and erosion, follows a hierarchical approach to deal with multiple scales and environmental contexts, and to integrate environmental, eco-engineering, and socio-ecological domains, establishing a balance between monitoring the engineering performance of NBS actions against landslides and erosion and the wider provision of ecosystem functions and services. The proposed framework also introduces aspects to encourage the upscaling process of NBS actions against the hazards under concern. This paper is structured as follows. In the first section, we introduce the study scope and context, as well as the case study in which this paper was framed. Then, we explore the rationale behind the proposed framework, and we outline its different compartments and dimensions. Next, we present the basis for identifying metrics and KPIs using the proposed framework within the established case study and with special emphasis on the monitoring stage of the NBS project. Finally, we discuss the novel aspects of the proposed framework, and its ability to build upon the NBS evidence base to encourage the uptake of NBS against landslides and erosion through facilitating and strengthening the process of monitoring NBS performance.

STUDY SCOPE, CONTEXT AND CASE STUDY

The scope of this study is to enable multidisciplinary teams of researchers, practitioners, and stakeholders to evaluate the multi-functional performance of NBS actions against landslides and erosion during the monitoring stage of the NBS project and against a pre-established baseline, providing a sound evidence base for NBS upscaling. To do so, we propose a novel framework helping to identify holistic sets of metrics and KPIs along the timeline of NBS project, providing a platform for monitoring the multi-functional performance of NBS against specific hydro-meteorological hazards (HMHs) - i.e., landslides and erosion, in a particular socio-ecological context (**Section 3**).

The proposed framework is adopted under the Open-air Laboratory established in the UK (OAL-UK) in the frame of the EU-funded Operandum project (OPEn-air laboRAtories for Nature-baseD solUtions to Manage hydro-meto risks; Finer et al., 2020) to investigate how co-created NBS can help in the management of shallow landslides and erosion. OAL-UK is located in Catterline, NE Scotland, where a series of slopes and cliffs rolling into the North Sea have been subjected to severe episodes of shallow landslides and of surface and coastal erosion in the past (**Figure 1A**). The action of the sea waves during past storm surges has contributed to the destabilisation of the toe of the slopes and cliffs. However, the two major HMHs considered in this study are mostly concerned with heavy rainfall episodes and the accumulation of surface water on the slopes and cliffs forming materials (Gonzalez-Ollauri and Mickovski, 2017a). The OAL-UK benefits from a local community who are both highly informed and accepting NBS. In 2012, after a major landslip in the village damaged property and road infrastructure, local residents formed the Catterline Braes Action Group (CBAG, https://www.cbag.org.uk/). CBAG has implemented NBS actions prior to their involvement with the Operadum project, approaching academic and industry experts with whom to co-create NBS. Co-creation of NBS at the OAL-UK has brought technical experts together with local authorities, local communities, and other end-users to collaborate on the definition, design, implementation and monitoring of NBS, for which effective communication avenues between the team members and with the project stakeholders had to be found and established to facilitate the co-creation process.

Following the major landslip event of 2012, CBAG were struggling to overcome barriers associated with contested ownership and responsibility for the slope which was restricting potential for either a public or private funded remedial project. As an alternative, they approached an academic team of researchers from Glasgow Caledonian

FIGURE 1 | (A) The OAL-UK is located in Catterline bay, in NorthEast Scotland, where **(B)** frequent landslide and erosion events are triggered by heavy rainfall and surface water accumulation, putting at risk properties and infrastructure in the village of Catterline; **(C)** left: live cribwall under construction and **(right)** after a dense vegetation cover has been established on the cribwall; **(D)** Left: vegetated slope grating under construction and **(right)** after the vegetation cover has started to get established on the NBS action. Credit photos: Albert Sorolla Edo–Naturalea.

University (GCU) comprising expertise in both physical and social sciences (fields spanning from environmental science, civil engineering, architecture, urban sustainability, geography, public engagement, and environmental psychology), with experience in both academia and industry. The team were interested in promoting a holistically understanding of the problem faced by the community in OAL-UK, and to propose multi-functional NBS actions, and to implement multi-dimensional, analytical approaches helping to break down the problem and solution into their basic elements. The project reflects a collaboration between the community (through CBAG) and academia reflecting a drive towards engaged research which seeks to bridge the academia/practice/community divide by seeking to extent relevance and impact of research by seeing practitioners and the community as not only beneficiaries of its outcomes but as a key part of the process, thus promoting mutual benefit by advancing both theoretical and practical knowledge in a real-world context. Engaged research promotes the co-production of knowledge and in this context the opportunity was presented to promote this strongly through co-creation across the project but especially during the design phase. This requires for researchers to be 'insiders' rather than objective researchers and for the community to act as active participants in the research process rather than the subjects. This provides the basis for an iterative research approach which is reflexive, and it is promoted by the mutual understanding and shaping of the research questions and consideration of its implications developing between the academics and the community.

The framework proposed herein (**Section 3**) represents the cumulation of the research outlined above and wider reflections from the expert team in working with NBS at OAL-UK. Since 2012, the research has followed an iterative approach to working on the OAL's slopes alongside its community, implementing small-scale geophysical interventions against landslides and erosion alongside well-considered and informed community engagement. The research team has previously published on this iterative approach, advocating for its ability to enable decision-making, widen the evidence base for NBS design and management, and promote mediation and collaboration (Mickovski and Thomson, 2018). These benefits are critical outcomes in a project striving for co-creation, where the participation of local and regional stakeholders is vital to then enhance the design and installation of NBS through their contextual knowledge and ensuring their acceptance of research that addresses landslides and erosion while providing socio-economical co-benefits (Anderson and Renaud, 2021). Engaged research facilitates co-creation between the academics and community promoting reflexivity within an iterative approach where small interventions are designed, implemented, monitored, and evaluated before the next intervention is progressed (Creswell and Creswell, 2018). This approach allows for adaptions to be informed by the outcomes of the previous intervention; researchers with the community can establish what worked, what didn't work, and make informed decisions on how to approach and improve the research based on these. Although iterative approaches are associated predominantly with the social sciences (Creswell and Creswell, 2018; Aspers and Corte, 2019), the authors have experienced success in the use of small, incremental interventions within the bio-geophysical landscape, finding it to have aided in the development of a deeper understanding of the physical characteristics of the OAL and therefore to apply NBS against landslides and erosion proactively (i.e., identify and prevent), rather than reactively (Mickovski and Thomson, 2018). For the socio-ecological aspects, an iterative approach can help ensure that the needs, priorities and expectations of the local stakeholders occupy a central place in the design and implementation of NBS. In a linear approach, stakeholders may only be significantly involved at the design and evaluation stages, limiting the opportunity for both of them to give input and for the research team to take this input into account. This provides a separation between the academic and community during the research, whereas an iterative approach provides local stakeholders with multiple opportunities to steer the direction of the interventions, as well as having the chance to see interventions succeed; this need for proof has been shown to be influential over acceptance and willingness to participate in NBS co-creation (Mickovski and Thomson, 2018; Anderson et al., 2021; Anderson and Renaud, 2021) as it builds trust in the NBS itself and in the research team. Therefore, the provision of "proof" of a successful intervention within an iterative approach benefits later cycles of co-creation of NBS interventions and actions against landslides and erosion.

The NBS actions against landslides and erosion identified and co-created for OAL-UK belong to the 'green infrastructure' typology (Eggermont et al., 2015) and in the third category of NBS proposed in the UICN's NBS framework (i.e. infrastructure; Cohen-Sacham et al., 2019), and they follow the principles of ground/soil bioengineering techniques (Schiechtl and Stern, 1996). Herein, we are focusing on two specific NBS actions against shallow landslides and erosion control - i.e., vegetated cribwall and live slope grating, (**Figures 1C,D**). Live cribwalls are retention walls built with timber logs which are deployed forming a crib that is then anchored to the slope and ground (**Figure 1C**). The crib is subsequently backfilled with earth materials and local vegetation (e.g., tree cuttings, saplings) is planted on the upper and external faces of the cribwall to provide long-term mechanical and hydrological stability (Gonzalez-Ollauri and Mickovski, 2017a). The slope above the cribwall is generally reworked and flattened (Gray and Sotir, 1996) and covered with vegetation. Slope gratings are slope 'skins' built with timber logs that form a lattice that is anchored into the slope (**Figure 1D**). The cells of the lattice are filled with earth materials and local vegetation is planted on the surface.

FRAMEWORK DESCRIPTION, DIMENSIONS AND COMPONENTS

We propose the 'rocket framework' (**Figure 2**) to help identify a holistic set of metrics and key performance indicators (KPIs) along the timeline of NBS projects seeking to manage and/or address context-specific hazards of landslides and erosion. The 'rocket framework' is a systems-based, heuristic framework (e.g., Eakin et al., 2017) that integrates multiple levels and domains resulting from undertaking a thorough system analysis (e.g., Calliari et al., 2019) by which we broke down 'analytically' the components and stages of projects concerning NBS against landslides and erosion. Each framework level corresponds to a stage along the NBS project timeline. Within each level of the framework, multiple, multi-dimensional compartments are integrated to help portray processes relevant for selecting, deploying, monitoring, and upscaling specific NBS actions against the HMHs of concern. The different dimensions of each compartment within the framework arise from the current definitions of NBS that feature in the peer-reviewed literature (e.g., Raymond et al., 2017; Ruangpan et al., 2020). As a result, each framework compartment unfolds into environmental, social, and economic domains, which were re-arranged into different dimensions depending on the project stage (**Figure 2**). The 'rocket framework' also contemplates the appearance of a new, emerging context resulting from deploying

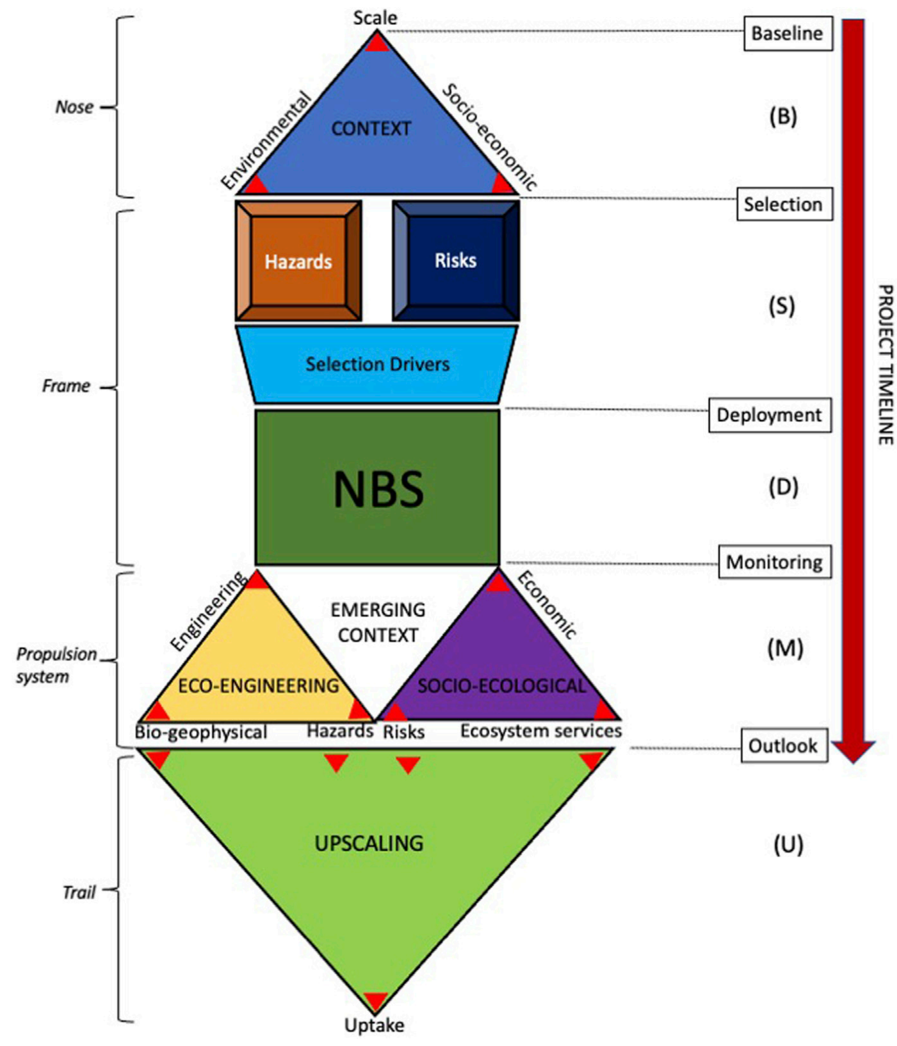

FIGURE 2 | The 'rocket framework' strives to facilitate the operationalisation of NBS actions against landslides and erosion by helping to articulate the thinking process involved in the identification of multi-functional key performance indicators at different stages of the NBS project -i.e., baseline establishment (B), NBS selection (S), deployment (D), and monitoring (M), which is the stage in which NBS performance against landsldies and erosion is envisaged to be essentially assessed. The vision of the 'rocket framework' is to 'uplift' the acceptance and reproduction of NBS actions through facilitating the provision of a strong evidence base on their performance. The 'rocket framework' is inspired in the illustration of spatial rockets (e.g., https://www.grc.nasa.gov/www/k-12/rocket/rockpart.html) to make it easy to memorise the dimensions and compartments that should be considered in the process of monitoring NBS against landslides and erosion. The capital letters in brackets help identify the project stage for the indicators provided in the repositories in **Table 1** and **Supplementary Material**.

NBS actions against landslides and erosion at a site which, through the correct functioning of the NBS, will be environmentally and socio-ecologically transformed and less prone to adverse natural disturbance and risks brought by landslides and erosion (Gonzalez-Ollauri and Mickovski, 2017b). A conceptual model illustrating the relationship between the multiple framework components is provided in **Figure 3**, which was used as the basis to identify groups of indicators (**Table 1** also see **Supplementary Material** for full description of metrics and their relationship with NBS performance). Although the main focus of this study rests in the monitoring stage of the project, which is the *propulsion system* of the "rocket framework" (**Figures 2**, **3**; **Table 1**), we are also providing a comprehensive set of metrics portraying the baseline, selection, deployment and upscaling of NBS against landslides and erosion (**Supplementary Material**). We believe that the latter metrics can also assist in the monitoring process to cast light on the performance of NBS.

Stage I: Project Baseline

The *nose* of the 'rocket framework' provides information about the context in which a given hazard and its related risks take place. This compartment belongs to the baseline stage of the NBS project, as it seeks to provide basic information on which the identification and selection of NBS actions are based upon. In addition, it strives to furnish basic information against which the performance of the selected NBS will be evaluated during the monitoring stage (**Figure 3**; **Table 1**). The context is portrayed by

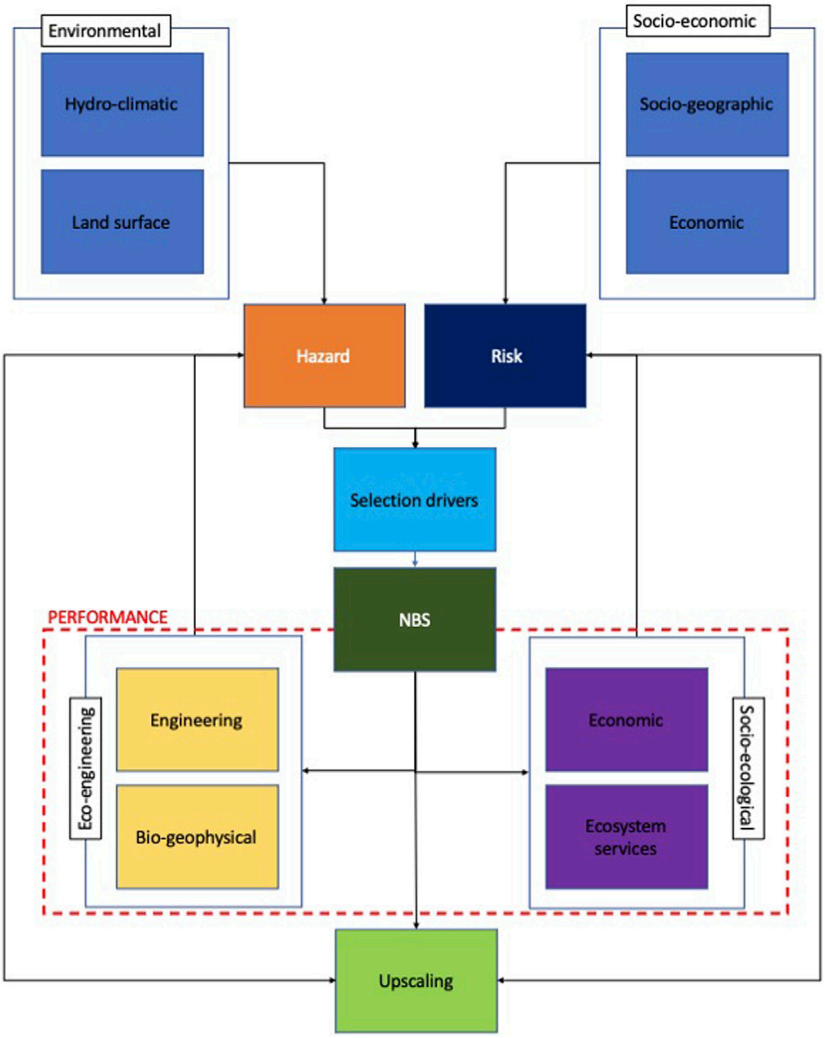

FIGURE 3 | Conceptual model illustrating the groups of indicators considered for each domain and compartment of the 'rocket framework' and their relationships throughout the NBS project timeline. The colour of the boxes is matching the colour assigned to the different project stages and domains within the framework (**Figure 2**). The groups featuring within the red dash-line frame belong in the monitoring stage, in which the NBS performance against landslides and erosion is envisaged to be essentially assessed.

the scale, and by its environmental and socio-economic dimensions. The environmental dimension of the context comprises attributes describing the hydro-climatic and land surface features of a site (e.g., air temperature, precipitation rate, topography, soil texture, land cover, etc; **Supplementary Material**). The environmental dimension also establishes the likelihood and recurrence of landslides and erosion at a given site -i.e., the problem for which NBS actions are designed in the context of this study. The socio-economic dimension of the context comprises social, economic, and cultural features of a particular site and is concerned with aspects such as population density, population demographics, economic activity, and cultural heritage (**Supplementary Material**). The socio-economic dimension establishes the boundaries within which citizen engagement and participation in NBS projects occur. The identification and engagement of stakeholders within the context is crucial to the function and success of NBS co-creation activities (Durham et al., 2014; Anderson et al., 2021). Here, a stakeholder mapping processes is established/envisaged to gain understanding of the interested groups: who they are, what their needs, expectations or priorities are, the extent to which their support is essential to aid project success, and the levels of interest and investment of certain parties (Durham et al., 2014; Talò, 2017). Through the understanding of these factors, effective stakeholder engagement strategies can be developed to ensure that the correct citizen groups are involved in the co-creation process at the correct time, and in an appropriate manner (Durham et al., 2014). Moreover, the socio-economic dimension determines the perceived virulence of landslides and erosion and the need for action against them: that is, the risk seen to be posed by these natural hazards. Studies have shown that risk is often a matter of perception, i.e. people who are

TABLE 1 | Indicators repository originating from the 'rocket framework' **(Figures 2, 3)** to monitor the performance of plant-based NBS against landslides and erosion at the monitoring stage (M).

Project stage	Domain	Compartment	Indicator number	Indicator[a]	Metric	Measurement unit	References
MONITORING (M)	Eco-engineering performance	Engineering	M1	Resistance to sliding	Active earth force	N m^{-1}	EN-1997-1
					Shear resistance	N m^{-1}	
			M2	Resistance to overturning	Active earth force	N m^{-1}	EN-1997-1
					Overturning resistance	N m^{-1}	
			M3	Resistance to shear failure	Wall-face contribution to stability	N m^{-1}	EN-1997-1
			M4	Resistance to bending	Bending stiffness	N m^{-1}	EN-1997-1
			M5	Pull-out resistance	Pull-out force	N m^{-1}	BS EN 1383:2016
			M6	Resilience and durability of the structure	Wood decay		EN 1995-1-1 Tardio and Mickovski (2016)
		Bio-geophysical	M7	Plant cover	Plant counts	No m^{-2}	
					Crown area	m^2 m^{-2}	Muukkonen and Makipaa (2006)
					Plant biomass	kg m^{-2}	Keeton (2008)
					Canopy cover fraction	%	Gonzalez-Ollauri and Mickovski (2017d)
					Plant mortality	No	
			M8	Plant growth	Height	m	Passioura (2002)
					Basal area	m^2	Gonzalez-Ollauri et al. (2020a)
					Crown area	m^2	
					Leaf area index	m^2 m^{-2}	
					Canopy cover fraction	%	
			M9	Plant diversity	Shannon index	-	Gonzalez-Ollauri and Mickovski (2017d)
			M10	Rainfall partitioning	Rainfall interception	mm	Zimmermann and Zimmermann (2014)
					Stemflow	mm	Gonzalez-Ollauri and Mickovski (2017a)
					Dripfall	mm	Gonzalez-Ollauri et al. (2020b)
			M11	Raindrops impact energy	Raindrop size	μm	Vaezi et al. (2017) Nanko et al. (2004)
			M12	Root profile	Root area ratio	%	Gonzalez-Ollauri and Mickovski (2016)
			M13	Root reinforcement	Root pull-out force	MPa	Gonzalez-Ollauri and Mickovski (2017b)
					Root tensile strength	MPa	Schwarz et al. (2010)
					Apparent root cohesion	kPa	Stokes et al. (2008)
			M14	Soil wetness	Volumetric soil moisture content	%	Lu and Godt (2010)
					Matric suction	kPa	Basist et al. (2006)
					Piezometric level	m	
					Soil wetness index		
			M15	Soil temperature		°C	Alvarez-Uria and Körner, (2007) Meyer et al. (2018) Gonzalez-Ollauri et al. (2020a)
			M16	Soil respiration	CO_2 efflux	μmol m^{-2} s^{-1}	Raich and Tufekcioglu, (2000)
			M17	Soil fauna	Soil fauna index	No	Yan et al. (2012)
					Species counts		Briones, (2014)
					Shannon Index		
			M18	Evapotranspiration		mm m^{-2} d^{-1}	Priestley and Taylor (1972)

(Continued on following page)

TABLE 1 | (*Continued*) Indicators repository originating from the 'rocket framework' **(Figures 2, 3)** to monitor the performance of plant-based NBS against landslides and erosion at the monitoring stage (M).

Project stage	Domain	Compartment	Indicator number	Indicator[a]	Metric	Measurement unit	References
					Potential evapotranspiration rate		Allen et al. (1998) Feng et al. (2020)
			M19	Belowground preferential flow	Bypass flow rate	mm s	Clothier et al. (2008) Zhang et al. (2018) Gonzalez-Ollauri et al. (2020b)
	Socio-ecological	Economic	M20	Energy use	Energy performance index (EPI)	kWh m^{-2}	UK Government – Department of Communities and Local Government, (2017) Bakar et al. (2015)
			M21	Water consumption	Water consumption Water exploitation index (WEI+)	m^3 m^{-2}%	Thomback et al. (2015) EEA, (2021)
			M22	Waste generation		Kg m^{-2}	Bakshan et al. (2015)
			M23	Whole life cost		£	
			M24	Operational cost		£	
			M25	Value of provided resources		£	Keesstra et al. (2018)
			M26	Employment generation		No. %	Wild et al. (2017)
			M27	Quality of construction	CONQUAS score	%	Low and Ong, (2014)
		Ecosystem services and co-benefits	M28	Accessibility of natural space		m2%	Frumkin et al. (2017)
			M29	Aesthetic perception		Qlt	Daniel, (2001)
			M30	Recreation	Organised recreation groups	No	Sutton-Grier et al. (2015)
					Informal recreation activities	Qlt	
			M31	Stakeholder engagement with co-creation		No	Soini et al. (2020)
			M32	Changes in well-being		Qlt	Van den Bosch and Sang, (2017)
			M33	Tourism generation		Qlt	Sutton-Grier et al. (2015)
			M34	Cultural heritage		m^2	Sutton-Grier et al. (2015)
			M35	Materials provided		No	de Groot et al. (2002)
			M36	Habitat provision		No m^2	Sutton-Grier et al. (2015)
			M37	Habitat support		No m^2	Sutton-Grier et al. (2015)
			M38	Air quality regulation	Daily Air Quality Index	Qlt	Defra, (2021)
			M39	Water cycle regulation	soil water mass balance		Voltz et al. (2018)
			M40	Carbon cycle regulation	Soil organic carbon Woodland carbon code	mg kg^{-1} tCO_2 ha^{-1}	Harden et al. (2018) Forestry Commission, (2018)
			M41	Climate regulation	Heat regulation index Moisture regulation index		West et al. (2011)
			M42	Landscape quality	Landscape quality indicator		Daniel, (2001) Sowińska-Świerkosz and Michalik-Śnieżek, (2020)

[a]*For full description of the metric and its relationship with NBS performance see Supplementary Material.*
Qlt., qualitative; No, number or counts.

exposed to the same hazard may not perceive their own or their community's level of risk equally (De Dominicis et al., 2015; Rufat et al., 2015). Risk perception has been shown to be influenced by emotions (e.g., fear), prior experience of hazards, trust in authorities and/or solutions, and place attachment (Keller et al., 2012; Wachinger et al., 2013; De Dominicus et al., 2015; Rufat et al., 2015).

The scale of the context can be both spatial and temporal. The consideration of the scale is crucial for establishing boundaries around a given context and to determine the size of the NBS action–e.g., landscape, catchment, stand, or individual intervention scale (e.g., Bock et al., 2005). Some HMHs can only be perceived within a given spatial and temporal scale (e.g., landslides–landscape scale and slow; erosion–both landscape and catchment scales, and both slow and fast; flooding–catchment scale and fast), and frequently recurring problems or hazards may require greater efforts and NBS actions that are more flexible and resilient to disturbance. It is also essential to consider both the temporal and spatial scales in the context of socio-economic considerations, as the scales will be a factor in the priorities and perceptions of stakeholders. Careful planning with the context's scale in mind is needed to ensure the connectivity between multiple NBS actions and the environment as well as the communities in which they are embedded, so they can perform effectively to deliver the functions for which they were designed (Calliari et al., 2019).

Stage II: Nature-based Solutions Action Selection

The *frame* of the 'rocket framework' comprises the selection and deployment (**Section 3.3**) stages of the NBS project (**Figure 2**). The selection stage firstly involves the characterisation of the hazard (i.e., landslide and/or erosion) and its associated risks, using metrics belonging to the environmental/geo-physical and socio-economic dimensions, respectively, which stem from the context compartment described above (**Figures 2**, **3**; **Supplementary Material**). In a co-creation approach, the involvement of stakeholders from the outset is a crucial feature of the selection stage. Following stakeholder mapping, the identified stakeholders make contributions to the definition and characterisation of the hazard, through providing their perception of it through informal conversation, public meetings, and focus groups. The selection stage should indeed reflect the on-site conditions and bio-geophysical evidence but also consider the priorities of the stakeholders, who must feel heard and represented to achieve successful co-creation (Talò, 2017; Anderson et al., 2021). There is a distinction to be made between hazard and risk: on one hand, the hazard under concern should be characterised in the light of direct, site evidence where possible (e.g. slow-moving hazards such as landslides) but, most likely, this is described in terms of its likelihood and recurrence using predictive and probability models (e.g., Gonzalez-Ollauri and Mickovski, 2017c). On the other hand, the perceived risk can be evaluated after the hazard's likelihood is known, and specific risk assessment frameworks and models can be used for this purpose (Shah et al., 2020). These normally involve using socio-economic variables (e.g., scale of property damage; scale of impact on economic activity; damage to cultural heritage; knowledge of hazard cause and prevention **Supplementary Material**) that are understood through analysis of risk perception of stakeholders (e.g., Shah et al., 2020). The characterisation of hazards and risks will inform the NBS selection process, and we suggest re-evaluating these two compartments at the monitoring stage in the frame of the emerging context (**Figures 2**, **3**; **Table 1**) to acknowledge whether the NBS actions are contributing to mitigate, manage, or reduce the hazard and the risks for which they were planned, and so NBS upscaling can be promoted (Cohen-Sacham et al., 2019).

The NBS selection process is characterised in the proposed framework (**Figure 2**) through a series of selection drivers associated to a co-creation process (**Supplementary Material**; Soini et al., 2020), which involves the participation of stakeholders mapped within the baseline definition stage. The selection drivers also belong in the environmental/bio-geophysical, and socio-economic dimensions, feeding a decision-making process in which NBS actions against landslides and erosion are first proposed by experts, they are then presented to participating stakeholders and assessed in terms of their feasibility and perception. This process involves a dialogue between those with technical expertise and the stakeholders, where a consensus is reached on NBS that are both effective against landslides and erosion in terms of bio-geophysical characteristics, and appropriate to meet the needs, expectations or priorities of the stakeholders (e.g., aesthetic qualities, cost, reduction of perceived risk). Once solutions are agreed upon the process moves to designing and deploying (or co-deploying) the selected NBS actions.

Stage III: Nature-based Solutions Deployment

The 'rocket framework' facilitates the strategic deployment of NBS actions against landslides and erosion at spatial locations where the hazards and risks have been identified, and where the implementation of a NBS action is feasible from the engineering, environmental and socio-economic viewpoint. The deployment stage of the NBS project is envisaged as an opportunity to promote further the participation of stakeholders (i.e., co-deployment), to exchange knowledge, and to build capacity in NBS science and practice across the private and public sectors. The NBSs identified herein (**Section 2**) require low machinery input and the utilisation of locally available resources, such as plant cuttings, timber logs, and earth materials, making it easier to engage with local communities (i.e., end-users) during the deployment process (e.g., http://www.efib.org/activities/).

Stage IV: Nature-based Solutions Performance Monitoring

The *propulsion system* of the 'rocket framework' comprises the monitoring stage of the NBS project (**Figures 2**, **3**). This stage strives to provide information about the performance of the NBS actions against landslides and erosion using KPIs from the eco-engineering and socio-ecological compartments (**Figures 2**, **3**; **Table 1**), to characterise the emerging context resulting from deploying NBS actions, and to 'uplift' the uptake of NBS against these HMHs across the private and public sectors through the provision of a robust evidence base. We understand that effective engagement with stakeholders together with the provision of co-benefits (e.g., ecosystem services, resilience towards further natural stress and disturbance, additional economic income; Raymond et al., 2017) are central in NBS projects, as these will foster the positive perception, acceptance, and upscaling of NBS (Cohen-Sacham et al., 2019). However, we wish to stress the importance of considering eco-engineering performance in NBS projects against landslides and erosion, as the socio-ecological performance of NBS actions will not be fulfilled as expected unless the NBS actions are delivering the ecological and engineering functions for which they were designed, thus managing and mitigating the HMHs under concern effectively and sustainably, and delivering an emergent, hazard and risk-free context.

The eco-engineering performance is herein concerned with the provision of tangible functions seeking to manage or mitigate landslide and erosion hazards. We believe that these functions can be quantified using engineering principles, which need input from the surrounding bio-geophysical environment. Moreover, the NBS action, understood here as a green infrastructure intervention, will transform the bio-geophysical context in which it is established, in turn regulating the engineering function of the NBS actions (e.g., Stokes et al., 2014). Consequently, the eco-engineering compartment comprises three dimensions: 1) engineering: evaluation of the engineering stability and resilience of NBS actions; 2) bio-geophysical: evaluation of the tangible changes triggered by NBS actions in the habitat, ecosystem and/or landscape in which they are established, and which are intrinsically related to the engineering functions the NBS actions perform; and 3) hazards: assessment of the likelihood and recurrence of landslides and erosion in the emerging context in which the NBS actions have been deployed.

The socio-ecological performance is chiefly concerned with the provision of additional goods and services to human communities (i.e., ecosystem services and co-benefits), rather than the provision of functions specifically related to managing landslides and erosion. These could be relating to an increase in access to the natural environment which, studies have shown, have positive impacts on physical and mental health (e.g., Frumkin et al., 2017). Increasing the provision of nature can also have economic benefits such as increasing the value of surrounding properties or increasing touristic income to an area (e.g., Trojanek et al., 2018; **Table 1**). Assessment approaches commonly used for quantifying ecosystem services bundles and synergies can be considered to assess co-benefits (e.g., de Groot et al., 2002; Gonzalez-Ollauri and Mickovski, 2017e). However, it is worth noting that co-benefits provided by a given NBS action can overlap with the eco-engineering functions, strengthening the interconnectivity of the elements of the emerging context following NBS implementation (**Figure 2**). Negative services or disservices should be considered, too (i.e., negative and unexpected impacts). For example, the vegetation cover established on the NBS can lead to the production of pollen, which may have a negative impact on public health. Additionally, increased touristic interest can become undesirable if the infrastructure services are not there to adequately support it (e.g., road capacity, parking, waste services). Care must also be taken in the selection of plant and seeds, to avoid the introduction of flora or fauna that would prove invasive to native species.

The proposed framework also considers economic and life-cycle aspects within the socio-ecological compartment (**Figures 2**, **3**). These aspects relate to the costs (time and money/carbon) and resources needed to conceptualise, procure, design, construct, operate/maintain/monitor and, in some cases, decommission the NBS (**Table 1**). Life-cycle assessment/analysis concepts (Klopffer and Grahl, 2014)) can be used to forecast the energy and material fluxes over the life cycle of the NBS and monetise them to the limits of their applicability (Ayres, 1995). In addition, the socio-ecological compartment also includes assessment of the risks under the conditions of the new emerging context, which can be re-evaluated following the same approaches used in earlier stages of the NBS project (**Figures 2**, **3**).

Stage V: Upscaling

The *trail* of the 'rocket framework' comprises the NBS upscaling stage, which is foreseen to be supported by the information generated in the monitoring stage (**Figures 2**, **3**; **Table 1**). The tip of the upscaling compartment is featured by the uptake of NBS actions by decision-makers and the public (Sarabi et al., 2020). Thus, the upscaling compartment contains a heterogeneous array of indicators focused on acceptability, perception, and well-being provided by NBS actions which could be framed as ecosystem services and/or co-benefits, but also, of bio-geophysical indicators transformed or regulated by the NBS actions and which can contribute to the reproducibility and future monitoring assessment of the upscaled NBS actions elsewhere and over time (**Supplementary Material**).

METRICS AND KEY PERFORMANCE INDICATORS FOR NATURE-BASED SOLUTIONS AGAINST LANDSLIDES AND EROSION

Definition, Identification, and Selection of Metrics and Key Performance Indicators

An indicator can be defined as a measure (metric) based on verifiable data that condenses complexity and conveys information (Haase et al., 2014). Herein, we refer to key performance indicators (KPI) to pool metrics able to provide information related to the performance of NBS actions against landslides and erosion during the monitoring stage of an NBS project. We also use the term 'key' because it is assumed that the indicator has undergone a selection process and, thus, the most representative metric for a given function/process has been selected under the existing constraints of the NBS project. For the other project stages (**Figure 2**), we use the terms metric or indicator instead of KPI, even though the identified metrics have also undergone a selection process, which should be refined further by future users of the framework on the basis of project context, scale, scope and capacity. Generally, there are multiple metrics available for one indicator or performance goal (**Table 1** and **Supplementary Material**), so metrics should be selected from the proposed pool in the light of the available skills and resources, scale of analysis, and/or feasibility to take measurements of the selected metric (Raymond et al., 2017). However, whichever metric is chosen, it should meet the principles of credibility, salience, legitimacy and feasibility (Cash et al., 2020). By following these principles for selecting metrics and indicators, a minimum level of comparability will be ensured between NBS and case studies, thus contributing to build upon the NBS evidence base (Kabisch et al., 2016). Yet, it is understandable and expected that the indicators will change with regards to context and scale (Raymond et al., 2017), as indicated above.

A wide range of metrics and indicators should be identified to reflect the multifunctionality of NBS actions against landslides and erosion (Calliari et al., 2019; **Figure 3** and **Table 1**). The proposed 'rocket framework' (**Figure 2**) provides, through its multiple compartments and dimensions (and their connections; **Figure 3**), a good basis to capture the multiple functions defining holistically the performance of NBS actions against the above-mentioned HMHs. To this end, each dimension in the framework can be understood as a gap that must be filled up with measurements to achieve a good level of insight into NBS performance against landslides and erosion during the monitoring stage (**Figure 3**; **Table 1**; also see **Supplementary Material** for full description of metrics and KPIs). Though the performance of the NBS should be fundamentally assessed during the monitoring stage, the project stages prior to monitoring will set a context and a baseline for testing the NBS performance.

In this study, the multidisciplinary team of researchers followed an analytical, brainstorming approach during a series of five meetings (i.e., one per NBS project stage) by which the problems and elements of the identified NBS actions (**Section 2**) were broken down into representative drivers, components, and expected outcomes, and by following the structure provided by the 'rocket framework' (**Figures 2**, **3**). This brainstorming approach, which sought co-creation between academic researchers of multiple disciplines, enabled the team's participation and access to the process, discourse and mutual understanding to reach consensual outcomes. In addition, engaging the OAL-UK's community in this process was necessary where relevant to ensure that the emerging outcomes reflect the context and preferences were gained from their local experience. It was also assumed that the identified NBS actions should manage landslides and erosion through the regulation of their drivers and components (e.g., Raymond et al., 2017). The variables and factors that are expected to be regulated by NBS actions against landslides and erosion will need measurement through monitoring to convey information on their performance, for which baseline information related to the context is also essential, as indicated above. Next, we carried out a quick scoping review of the peer-reviewed scientific and grey literature (e.g., Collins et al., 2015), which was not intended to be comprehensive but informative enough to identify metrics for each indicator. The scoping consisted in three stages (Raymond et al., 2017): 1) structured search of the peer-review scientific, including textbooks, and grey literature, including standards, using Google Scholar, 2) selection of literature resources based on relevance to problem and/or specific indicator, and 3) narrative synthesis of the selected scientific literature. In total, 99 documents were read to at least the abstract level (**Table 1** and **Supplementary Material**).

Context Indicators

Context indicators provide information about the baseline on which NBS actions against landslides and erosion are established (**Supplementary Material**; **Figure 2**). The problem of landslides and erosion can be described on the basis of context indicators referring to the scale and environmental factors underpinning the problem. The extent and risks associated to it can be characterised using socio-economic attributes (**Supplementary Material**). The context indicators are not classified as KPIs herein, as they are not explicitly referring to the NBS performance. Yet, NBS performance should be assessed based on its context or baseline.

Following the three dimensions established in the context compartment of the 'rocket framework' (**Figure 2**), and following the analytical, brainstorming approach outlined above, we identified a series of features that enabled us to describe the context and establish a baseline for our case study (**Section 2**). An extensive but not exhaustive list of context indicators and related metrics is shown in **Supplementary Material**. Regarding the scale, both space and time were considered in order to establish a baseline related to the geographic size of the landslides and erosion events occurring at OAL-UK, which was set at the landscape scale, as well as its frequency and recurrence (Mickovski and Thomson, 2018). Regarding the environmental dimension, hydro-climatic and land surface features were considered to be relevant for understanding and predicting landslides and erosion events (**Figure 3**; **Table 1**; Gonzalez-Ollauri and Mickovski 2016; Gonzalez-Ollauri and Mickovski, 2017c). The socio-economic dimension was divided into socio-geographic and economic domains (**Figure 3**; **Supplementary Material**). The metrics within socio-economic context are primarily described by secondary source data from the Scottish Index of Multiple Deprivation (SIMD) 2016 and Scottish Census 2011, both of which also establish the spatial scale as data output zones (**Supplementary Material**). The SIMD and Census gather socio-economic data at a national level every 4 and 10 years respectively, allowing local, regional and national comparisons to be drawn. The most relevant dimensions were deemed to be population size and demographics (i.e., gender, age, education and employment levels) in addition to geographic dimensions such as access to and services and infrastructure.

Indicators for the Hazards, Risks, and Nature-based Solutions Selection and Deployment

Hazard indicators are those conveying information about the occurrence of a landslide and/or erosion event *ex-ante* (i.e., before the NBS action has been deployed) and *ex-post* (i.e., after the NBS has been deployed). Hazard indicators were split into predictive and empirical (**Figure 3**; **Supplementary Material**). Predictive hazard indicators strive to provide robust information about the likelihood and recurrence of the landslide and/or erosion event. These indicators are data- and computationally-intensive, as their calculation depends on the availability of relevant time series, as well as to data from a comprehensive set of variables, which are normally related to the bio-geophysical context. These indicators can be based on statistical modelling (e.g., Gonzalez-Ollauri and Mickovski, 2017c), which can evaluate the probability of a landslide and/or erosion event on the basis of a baseline feature, such as rainfall intensity or runoff, or they can be based on more elaborated, process-based indices combining multiple variables from the context, such as the soil loss equation for computing erosivity (Benavidez et al., 2018), or the limit equilibrium model for computing slope stability (Lu and Godt, 2013). Empirical hazard indicators provide first-hand evidence about a particular hazard and they must be collected on site or using primary data. For the case of landslides and erosion, it is convenient to follow principles and protocols from geotechnical engineering (e.g., AGS, 2007) and edaphology (e.g., Morgan, 2004). Examples of empirical indicators for these two hazards are those providing information related to soil mass movement and deformation or to land exposure (**Supplementary Material**).

Risk indicators are defined by not just the context and hazard that they are relating to, but also by the perception of the stakeholders who experience the hazard (**Figure 3**). Consequently, risk indicators were split into those relating to 'damage' and those relating to 'risk perception' (**Supplementary Material**; also see **Supplementary Material** for full description of selection and deployment metrics). As previously discussed, factors such as prior experience of hazards, or knowledge of (and preparedness to respond to) a hazard can increase or reduce the level of risk a population perceives themselves to be at (De Dominicis et al., 2015; Shah et al., 2020). Risk indicators are therefore both objective–in that there is often a measurably likelihood that a hazard will place a population under a certain risk–and subjective, as a population can perceive a risk as less when they are accepting, aware or prepared for it (e.g., De Dominicis et al., 2015). We followed herein the approach proposed in Shah et al. (2020), where the risks within NBS sites are broadly categorised by four factors: 1) ecosystem susceptibility–indicators can be biodiversity levels, rate of shoreline erosion; 2) ecosystem robustness–indicators can be presence of environmental protection policies, hardiness of agriculture and biodiversity; 3) social susceptibility–indicators can be diversity in sources of economic income; presence of natural and cultural heritage protection; property values and insurance costs; and 4) coping and adaptive capacity–indicators can be presence of protective measures against hazard; monitoring systems, community action plan against hazard.

NBS selection indicators are informed both by the bio-geophysical characteristics of the site, and by the needs, expectations and priorities that emerge from stakeholder mapping and engagement processes (**Figures 2**, **3**). NBS selection indicators emerging from stakeholders can be aesthetic perception (e.g., increasing or preserving the natural aesthetic of their community), installation and maintenance costs, or speed and visibility of results. Consequently, we divided the selection drivers for NBS against landslides and erosion into four groups: 1) hazard-specific, 2) site-specific, 3) economic, and 4) socio-ecological (**Supplementary Material**; **Figure 3**). Similarly, we split NBS deployment indicators into socio-ecological, engineering, and bio-geophysical domains (**Supplementary Material**; **Figure 3**) with the aim to provide an integrated picture of the factors that may affect the eventual deployment of NBS actions following the selection process (**Supplementary Material**).

Eco-Engineering Indicators

Indicators conveying information related to eco-engineering functions can be envisaged after the context and problem have been described with context indicators (**Section 4.2**). We assumed that the NBS action will contribute to manage and/or to regulate those drivers and variables triggering and influencing landslides and erosion. Consequently, the eco-engineering performance of the NBS action can be quantified through the assessment of these drivers and variables during the monitoring stage (**Figure 3**; **Table 1**; also see **Supplementary Material** for full description of eco-engineering indicators). For the case of NBS actions against landslides and erosion, eco-engineering indicators should inform on how the NBS actions contribute to regulate the hydro-climatic and land surface indicators (**Figure 3**), constituting the set of bio-geophysical indicators contributing to eco-engineering performance (**Table 1**), thus being classified herein as KPIs. The indicators portraying the engineering performance of the NBS, which are also classed as KPIs, can be established on the basis of the internal stability and resilience/durability of the NBS action or structure, supplemented with hazard-specific indicators, which they can be assessed using geotechnical engineering principles (Jones, 1996).

The identification and subsequent selection of metrics for the pool of identified eco-engineering KPIs was undertaken on the basis of reviewing textbooks and manuals for standard civil/geotechnical engineering practice (e.g. Eurocode Standards EN-1997-1; Jones, 1996), from which one can gain insight into the principles of slope stability and protection to manage landslides and erosion problems, and into the mechanisms and mathematical principles by which retention walls (e.g., cribwall; **Figure 1C**) and slope 'skins' (e.g., slope grating; **Figure 1D**) contribute to the management of the hazards under concern (e.g., Gray and Sotir, 1996). Once the key metrics were identified, we proceeded with the quick scoping process of the peer-review literature, to identify metrics by which the living component of the NBS (i.e., vegetation) can contribute to regulate these metrics (e.g., Norris et al., 2008). The collection of metrics for each eco-engineering KPI is gathered in **Table 1**. The scoping process also helped to set/propose thresholds for each quantitative metric (**Table 1** and **Supplementary Material**), which were established herein on the basis of metrics' values worsening the occurrence of landslides and erosion, or affecting negatively to the NBS performance. However, we think that the creation of compound, performance indicators supplemented with sensitivity analyses (i.e., break-point analysis) can help to elucidate indicator thresholds (e.g., Toms and Lesperance, 2003; **Section 5**).

Socio-Ecological Indicators

Insights into the socio-ecological performance of NBS actions against landslides and erosion are of the utmost importance to evaluate the overall performance of NBS and, more importantly, to promote their public acceptance, upscaling, and reproduction (Saleh and Weinstein, 2016; Raymond et al., 2017; Lafortezza et al., 2018). Consequently, socio-ecological indicators were classified as KPIs. The socio-ecological performance has to be measured using multiple qualitative methods of assessment such as focus groups, surveys, and observations. The establishment of baselines and thresholds for socio-ecological KPIs is often more challenging than for eco-engineering KPIs, as they are more intrinsically linked to not only the demographic and socio-economic characteristics of the community under question, but are influenced by the needs, expectations and priorities that emerge through the stakeholder mapping and engagement processes (see **Sections 2**, **3.1** and **3.2**; Durham et al., 2014; **Table 1**). These subjective matters are further compounded when considering the upscaling of NBS (**Section 4.6**); a large scale NBS project may contain multiple communities with differing socio-economic profiles, and stakeholder priorities, and therefore require a carefully considered approach to measuring KPIs at both the micro and macro scales.

Socio-ecological KPIs are partially informed by socio-economic and ecological metrics, and partly through the stakeholder mapping and engagement process (**Section 3.1**; **Table 1**; also see **Supplementary Material** for full description of socio-economic indicators). They relate to the ecosystem services and co-benefits associated with the NBS actions, with the costs and benefits related to the intervention and with the site-specific risks encountered in the emerging context (**Figures 2**, **3**; **Table 1**). Socio-ecological KPIs can thus include those directly resulting from the NBS, such as the public accessibility to natural spaces and the perceived aesthetic quality of the community (Sutton-Grier et al., 2015; Keesstra et al., 2018; **Table 1**), or the regulation of the water cycle at the landscape level or the promotion of plant diversity and soil fauna (Keesstra et al., 2018; **Table 1**), as well as those indirectly related to the NBS, such as benefits to physical and mental health, the increase in employment opportunities, the increase in property value, or avoidance of damage costs (van den Bosch and Sang, 2017; Wild et al., 2017; **Table 1**). There are also socio-ecological KPIs relating to community cohesiveness through an increase in stakeholders involved in hazard mitigation projects, which could be classified as the provision of cultural value and heritage by the NBS (Keesstra et al., 2018; **Table 1**). To assess the economic performance of a NBS action against landslides and erosion, we identified financial KPIs feeding into cost-benefit analyses (e.g., Vicarelli et al., 2016), comparing, for example, whether the life cycle costs of a NBS action would be lower than those of a traditional 'grey' solution because of the absence of structural concrete and steel, the use of natural materials, lower maintenance costs and the carbon footprint offset of the construction provided by the used vegetation.

Upscaling Indicators

To scale up NBS actions against landslides and erosion, it is essential to provide evidence during the monitoring stage to build confidence in NBS and promote their uptake by the public and private sectors, encouraging decision and policy makers to include NBS in their agendas (Sarabi et al., 2020). To do so, we believe that four main fronts or dimensions across the socio-ecological and eco-engineering compartments need assessment during the monitoring stage of the NBS actions (**Figures 2**, **3**; also see **Supplementary Material** for full description of upscaling metrics), from which upscaling metrics can be retrieved (i and ii) ecosystem services and risk perception: it is essential to demonstrate with supporting stories and examples how NBS actions are able to provide multiple benefits and co-benefits to human communities whilst contributing to reduce risks and changing the perception towards them by exposed and vulnerable communities (iii and iv) hazard mitigation and bio-geophysical environment: it is also essential to prove that specific NBS actions are in fact able to provide the functions for which they were designed and thus contribute to manage and mitigate landslides and erosion through the positive transformation of the bio-geophysical environment in which they were deployed. The latter would provide valuable evidence on the ability of NBS actions to promote climate adaptation, which is a key issue to reach global movements for NBS (IEEP, 2020). It has been established that local communities are more accepting of NBS–and more willing to participate in their deployment - when they have tangible evidence of the ability of it to prevent or significantly reduce impacts from HMHs (Anderson et al., 2021). Ergo, the evidence of effective mitigation of landslides and erosion through NBS could not only provide a scientific evidentiary basis to support the upscaling of NBS, but also create NBS advocates within communities to drive this upscaling.

DISCUSSION AND CONCLUSION

We proposed a systems-based framework that captures heuristically and holistically the complexity of the context in which NBS actions against landslides and erosion are established. The latter strives to facilitate the monitoring process of NBS performance over time with multi-functional KPIs (**Figures 2**, **3**; **Table 1** and **Supplementary Material**) together with context, selection and upscaling metrics and indicators (**Supplementary Material**), which were identified through a process of system analysis stimulated by the framework. We thus believe that the proposed framework can have a positive impact on the operationalisation of NBS actions against landslides and erosion, and on the establishment of an evidence base supporting future upscaling activities.

The 'rocket framework' (**Figure 2**) can help to provide a simplified, yet integrated, portrait of the landscape in which NBS against landslides and erosion are deployed, and to facilitate monitoring of the multi-functional performance of NBS by incorporating the relationships and feedbacks between social, economic, environmental, and engineering components, connecting the socio-ecological and bio-geophysical components of risk (**Figure 3**; **Table 1**; Gardner and Dekens, 2007). The 'rocket framework' is dynamic, as it interconnects the project stages to assess hazards and risks under changing, emerging contexts resulting from the functions and services provided by NBS actions over time (**Figures 2**, **3**; **Table 1**). The latter supports the implementation of adaptive management strategies in the event of unsatisfactory NBS performance against landslides and erosion (Cohen-Sacham et al., 2019). The 'rocket framework' is flexible, as it is generic enough to incorporate different pools of indicators than the proposed herein (**Table 1** and **Supplementary Material**) to meet the needs of different contexts and challenges (i.e., different HMHs than landslides and erosion), and it is also versatile, as it can be used at different project stages to identify problems, compare alternatives, or monitor performance of established NBS.

A novel key aspect of the 'rocket framework' is that it integrates, for the first time, components related to eco-engineering performance in a NBS framework tailored to landslides and erosion (**Figure 2**; **Table 1**). We think that this is essential when NBS actions focus on infrastructure necessitating the intrusive intervention of the ecosystem/ landscape to address the challenge under concern, as is the case for landslides and erosion. NBS are planned to solve a specific problem (or a series of them), so it is essential to be able to quantify how well a given NBS is doing with solving the problem under concern. The eco-engineering domain clearly established the scope and objective of the selected NBS actions detailed in the case study - e.g., slope stability and ground protection with live cribwall and vegetated slope grating (**Figures 1C,D**). It also provided a platform with a series of tangible, standard measures to quantify the effect of the NBS actions against the identified problems (**Table 1**). Additionally, the eco-engineering domain helped articulate the thinking process overarching other domains and dimensions depicted in the 'rocket framework', thus facilitating the identification process of KPIs (**Table 1**). The engineering performance of green infrastructure interventions is often taken for granted, as it is based on rigorous design and planning. As a result, this domain is often excluded from the pool of functions and benefits that NBS can provide. However, the eco-engineering domain may help envision how technology and nature blend together to provide solutions for specific challenges and to provide benefits to the society. Building upon the case study explored herein (**Section 2**), we provide an example to cast some light on how the eco-engineering domain of the 'rocket framework' helped to articulate the identification and selection of KPIs for the selected NBS actions against landslides and erosion (**Table 1** and **Supplementary Material**).

The chances of landslides and erosion events will be substantially reduced under flat topographies (e.g., Panagos et al., 2015), under well-structured, well-reinforced and relatively dry soil (Lu and Godt, 2013), and under an ecosystem/landscape that is resilient to change and disturbance brought by the hazards (Walker, 2013; Gonzalez-Ollauri and Mickovski, 2017c). Thus, the selected NBS actions against landslides and erosion should potentially modify the site topography by reworking and flattening the slope where they are deployed (i.e., re-grading; Norris et al., 2008). The timber structure of the NBS actions (**Figures 1C,D**) and their living components (i.e., plants) should reinforce mechanically the ground either through the insertion of new structural elements in the soil such as timber members, steel/wooden nails or plant roots (e.g., Jones, 1996; Gonzalez-Ollauri and Mickovski, 2017d), or through, for example, the long-term incorporation of organic matter to the soil originating in the decay of plant parts (e.g., Adamczyk et al., 2019). The establishment of a dense vegetation cover on the NBS structure (**Figure 1C**) should promote drainage and water uptake, overall leading to drier soil conditions (Gonzalez-Ollauri and Mickovski, 2017b; Gonzalez-Ollauri and Mickovski, 2020a) and to the regulation of the local climate (Osborne et al., 2004). Moreover, the establishment of the vegetation cover on the NBS will contribute to intercept rainfall (Gonzalez-Ollauri and Mickovski, 2017b), to reduce the mechanical impact of raindrops on the soil (Vaezi et al., 2017), to regulate the temperature in the soil (Gonzalez-Ollauri et al., 2020b), to stimulate the colonisation by soil fauna and native flora, and much more (e.g., hosting pollinators and birds; seed dispersal, pest regulation, resistance to windstorms, etc. Brockerhoff et al., 2017); providing overall resilience towards change and disturbance and making the ecosystem more complex and stable (Pimm, 1984). Plant establishment and development will make the intervened landscape aesthetically pleasant (Smardon, 1988), encouraging recreational activities within the intervened area, such as walks or birdwatching (Shanahan et al., 2015), and fostering the positive perception and acceptance of the NBS actions by the human communities exposed to landslides and erosion such as the community at OAL-UK (**Section 2**); provided that effective communication and engagement with the end-users is established to increase their awareness of the benefits (Anderson and Renaud, 2021). The stabilisation of the slope with a solid, timber structure that eventually merges with the local landscape (**Figures 1C,D**) will also have a positive impact on the risk awareness and perception by the affected community.

The example provided above illustrates the connection between the eco-engineering and socio-ecological domains established in the 'rocket framework' (**Figures 2**, **3**) in a context of landslides and erosion management and mitigation. It also draws an example about the thinking process by which additional domains, compartments, and indicators unfolded through the critical analysis of the system, problems, and solutions, using the eco-engineering domain as driver. The latter stresses the value of including the eco-engineering domain in the monitoring process of NBS performance against landslides and erosion, as it allows envisioning how technology and nature blend together to provide solutions for specific challenges. Thus, we believe that the 'rocket framework' and its associated analytical approach, by which it was conceived and

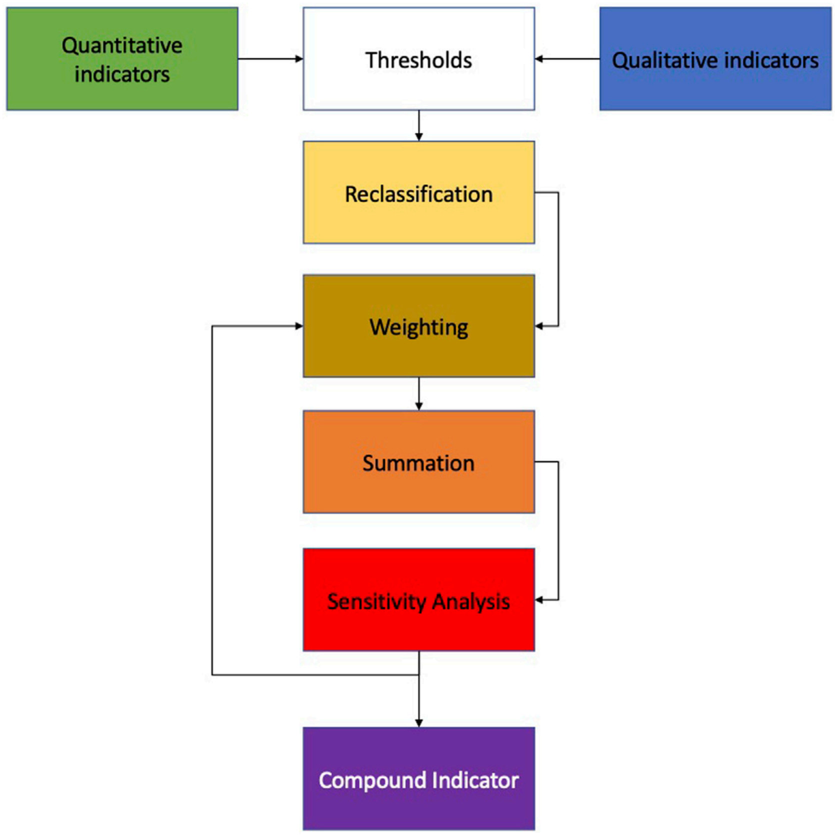

FIGURE 4 | Sequence of steps envisaged to generate compound indicators for NBS performance against landslides and erosion using the repository of indicators defined with the 'rocket framework'.

supplemented, can provide a good basis for the operationalisation of NBS actions against landslides and erosion, and for the quantification of their multi-functional performance through monitoring activities. It is worth noting that although the 'rocket framework' was conceived in a context of landslides and erosion, its dimensions and components were defined from a generic standpoint to enable reproducibility in other contexts and with different HMHs. However, it was beyond the scope of this study to provide the reader with a method to compute the overall performance of the NBS actions with the 'rocket framework' (**Figure 2**) and the KPIs and metrics identified (**Table 1**). The combination of numerical modelling, multi-criteria (MCA) and cost-benefit analyses is generally proposed in the literature to undertake such a task (e.g., Raymond et al., 2017). However, only few studies have attempted to combine multiple KPIs in the context of NBS performance to then provide a system of NBS scores or grades (e.g., Watkin et al., 2019). Hence, future studies should strive to address this gap by proposing and validating robust, numerical approaches that combine multiple quantitative and qualitative variables from the KPIs repository with the aim of producing a compound index or score conveying reliable information on NBS performance against landslides and erosion. We envisage that such approaches should at least involve the following four steps stemming from MCA (**Figure 4**):

1) reclassification: to change the values of one variable into other values, putting different variables on the same scale. With this step, a new score scale can be established for reclassifying the values of a given variable/indicator into intervals or groups. This process becomes easier when thresholds for a given indicator are identified (**Table 1** and **Supplementary Material**). Reclassification can also be useful to transform qualitative into numerical variables.
2) weighting: to allocate a measure of importance to the different variables or indicators involved in calculating NBS performance against landslides and erosion. It could be assumed that all the indicators are equally important but, most likely, some indicators are more relevant than others upon determining NBS performance. The weighting process can be supplemented with correlation and sensitivity analyses and/or with regression modelling when enough data are available, so only uncorrelated indicators are considered to compute the NBS performance score, and so trade-offs and synergies between indicators can be detected (Gonzalez-Ollauri and Mickovski, 2017e). Expert-driven techniques, such as the analytical hierarchy process (Saaty, 1980), can help identify objectively the relative importance of the different indicators involved (Gonzalez-Ollauri et al.,

2020b), but it is still important that only uncorrelated, independent indicators are taken to the next step.

3) summation: to combine the multiple indicators together and calculate the NBS performance score; once the indicators have been standardised, reclassified, and different weights allocated to each of them. The most widely approach to do so is the simple additive weighting (SAW; Hwang and Yoon, 1981). Yet, machine learning algorithms such 'boosted regression trees' (Breiman et al., 1984) and 'random forest' (Breiman, 2001) may open-up an exciting opportunity to combine multiple indicators, whether raw or processed, whether qualitative or quantitative, to retrieve scores or indices of NBS performance against landslides and erosion.
4) sensitivity analysis: to assess how the uncertainty in the NBS performance score can be allocated to the different indicators used and dismiss those indicators that do not significantly contribute to the output. If the uncertainty of the performance score is high, uncertainty filtering techniques can be implemented (e.g., Malkawi et al., 2000). Also, this step can help identify indicator thresholds through break-point analysis (e.g., Toms and Lesperance, 2003).

There is a pressing need to work along with nature to sustainably address current and future societal challenges that stem from environmental and climate change (e.g., EU Strategy on Green Infrastructure). Nature-based solutions against landslides and erosion open-up an exciting opportunity to do so, but they need upscaling, so their effect can be noticeable (Cohen-Sacham et al., 2019). There is, however, a severe lack of evidence on NBS performance (e.g., Nelson et al., 2020; Ruangpan et al., 2020) which hinders the operational rigour of NBS, it undermines the trust society has in them, and it slows down the upscaling and overall uptake of NBS. Filling the knowledge gap on NBS performance against landslides and erosion is an ambitious challenge that will require the close cooperation between scientists, practitioners, end-users, human communities, and decision and policy makers. In this study, we are providing a novel holistic framework based on the experience from a relevant case study that strives to facilitate addressing the lack of NBS performance evidence against landslides and erosion by helping to articulate the thinking process involved with mapping out effective monitoring strategies throughout the project timeline, thus helping identify problems, solutions, and performance indicators holistically. Herein, we are also refocusing the spotlight towards green infrastructure and eco-engineering techniques, which hold valuable experimental practice and knowledge to help build upon the evidence base on NBS against landslides and erosion, and their subsequent standardisation. This research showcases the benefits of engaged research which represents collaboration between a multidisciplinary academic research team which is seen as essential to help to shape the holistic coverage of the indicators, and also the co-creation process with community stakeholders at OAL-UK, which was deemed essential for ensuring local context is reflected and in gaining buy in through a shared mutual benefit between academics and the community on a theoretical and practice-based level. An iterative approach which promotes inclusion of actors and enables reflexivity throughout is deemed key to helping promote the conditions for co-creation. Future work will showcase the implementation of the proposed framework and KPIs repository in the OAL-UK, from which a reproducible approach to score NBS performance against landslides and erosion will be devised.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author.

AUTHOR CONTRIBUTIONS

AG-O: Conceptualisation, Data curation, Investigation, Methodology, Project administration, Visualization, Writing–original draft, Writing–review and editing. KM: Investigation, Methodology, Writing–original draft. CT: Methodology, Writing–original draft. SM: Supervision, Writing–original draft, Writing–review and editing. RE: Funding acquisition, Supervision, Writing–review and editing.

FUNDING

This work was carried out under the OPERANDUM (OPEn-air laboRAtories for Nature baseD solUtions to Manage hydro-meteo risks) project, funded by the EU's Horizon 2020 research and innovation programme (GA No: 776848).

ACKNOWLEDGMENTS

The authors appreciate the ongoing participation and interest of the community in Catterline, Scotland, in Nature-based Solutions projects.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/feart.2021.676059/full#supplementary-material

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frontiers in Environmental Science

SYSTEMATIC REVIEW
published: 22 November 2021
doi: 10.3389/fenvs.2021.737659

Nature-based Solutions in Bangladesh: Evidence of Effectiveness for Addressing Climate Change and Other Sustainable Development Goals

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OPEN ACCESS

Edited by:
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Specialty section:
This article was submitted to Interdisciplinary Climate Studies, a section of the journal Frontiers in Environmental Science

Received: *07 July 2021*
Accepted: *20 October 2021*
Published: *22 November 2021*

Citation:
Smith AC, Tasnim T, Irfanullah HM, Turner B, Chausson A and Seddon N (2021) Nature-based Solutions in Bangladesh: Evidence of Effectiveness for Addressing Climate Change and Other Sustainable Development Goals. Front. Environ. Sci. 9:737659. doi: 10.3389/fenvs.2021.737659

Many lower-income countries are highly vulnerable to the impacts of natural disasters and climate change, due to their geographical location and high levels of poverty. In response, they are developing climate action plans that also support their sustainable development goals, but conventional adaptation approaches such as hard flood defenses can be expensive and unsustainable. Nature-based solutions (NbS) could provide cost-effective options to address these challenges but policymakers lack evidence on their effectiveness. To address this knowledge gap, we focused on Bangladesh, which is exceptionally vulnerable to cyclones, relative sea-level rise, saline intrusion, floods, landslides, heat waves and droughts, exacerbated by environmental degradation. NbS have been implemented in Bangladesh, but there is no synthesis of the outcomes in a form accessible to policymakers. We therefore conducted a systematic review on the effectiveness of NbS for addressing climate and natural hazards, and the outcomes for other sustainable development goals. Research encompasses protection, restoration and participatory management of mangroves, terrestrial forests and wetlands, as well as conservation agriculture and agro-forestry, but there is an evidence gap for urban green infrastructure. There is robust evidence that, if well-designed, these NbS can be effective in reducing exposure to natural disasters, adapting to climate change and reducing greenhouse gas emissions while empowering marginalized groups, reducing poverty, supporting local economies and enhancing biodiversity. However, we found short-term trade-offs with local needs, e.g. through over-harvesting and conversion of ecosystems to aquaculture or agriculture. To maximize NbS benefits while managing trade-offs, we identified four enabling factors: support for NbS in government policies; participatory delivery involving all stakeholders; strong and transparent governance; and provision of secure finance and land tenure, in line with international guidelines. More systematic monitoring of NbS project outcomes is also needed. Bangladesh has an opportunity to lead the way in showing how high quality NbS can be deployed at landscape scale to tackle sustainable development challenges in low to middle income countries, supporting a Green Economic Recovery. Our evidence base highlights the value of protecting irreplaceable natural assets such as mangroves, terrestrial forests and wetlands, and the non-market benefits they deliver, in national planning policies.

Keywords: nature-based solutions, Bangladesh, climate change adaptation, climate change mitigation, resilience, sustainable development goals, governance, climate policy

INTRODUCTION

Many lower-income countries are highly vulnerable to natural disasters and climate change (Chen et al., 2015; Eckstein et al., 2019). As well as being in geologically and/or hydrodynamically unstable areas, and subject to extreme weather, their adaptation options are often limited by low financial, manufactured and human capital, the latter due to low levels of education and healthcare (Moser and Ekstrom, 2010; Spires et al., 2014; Shi et al., 2016). In response, many are developing National Adaptation Plans and Nationally Appropriate Mitigation Actions that seek to adapt to climate change, reduce disaster risk and cut greenhouse gas (GHG) emissions whilst also supporting the delivery of other sustainable development goals (SDGs). Yet commonly adopted adaptation and development approaches such as hard flood defenses (Narayan et al., 2016; Reguero et al., 2018; Ware et al., 2020) and intensive agriculture (Rasul and Thapa, 2004; Prabhakar, 2021) can be expensive and unsustainable. These interventions are static, so that they can become obsolete as climate threats intensify, and often tackle one problem whilst making others worse, for example by increasing GHG emissions and polluting water supplies (Rasul and Thapa, 2004; Prabhakar, 2021). Nature-based solutions (NbS) offer a more holistic approach to societal challenges, by working with and enhancing the natural, human and social capital that underpins long-term human wellbeing. NbS, either alone or combined with other approaches, could thus contribute to cost-effective options for addressing climate change, natural hazards and development challenges while also reversing biodiversity loss (Seddon et al., 2020). However, integration of NbS into national policies is limited (Seddon et al., 2019), partly because policy-makers lack accessible information on their effectiveness for delivering these benefits. The evidence that exists is dispersed across academic papers in journals from the physical, natural and social sciences (Chausson et al., 2020; Seddon et al., 2020), often behind paywalls, or buried in ‘grey literature’ reports scattered across many different websites. This presents a barrier to policymakers with limited time and resources.

To address this, we have compiled a comprehensive and accessible synthesis of evidence on the effectiveness of NbS for addressing climate change in Bangladesh, one of the most vulnerable countries in the world to the impacts of natural and climate disasters, which are compounded by environmental degradation and socio-economic challenges (Shi et al., 2016). Cyclones, which are becoming more intense due to climate change (Kossin et al., 2020), cause wind damage, coastal flooding and erosion, and together with sea level rise this contributes to more extensive storm surges (Hoque et al., 2019). The resulting inundation leads to salinization of soil and groundwater that destroys agricultural livelihoods (Wicke et al., 2013; Imam et al., 2016), while saline intrusions are also exacerbated by over-extraction of groundwater for irrigation (Zahid et al., 2018). In hilly areas such as the Chittagong region, the combined effects of forest degradation, hill cutting for housing construction and severe rainfall events cause soil erosion and landslides (Islam and Rahman, 2019), which can be triggered by earthquakes. Climate change is also leading to more severe heat waves and droughts (Imam et al., 2016), while vulnerability to water scarcity is worsened by high levels of water pollution, including widespread pollution of groundwater by arsenic (Akhter and Uddin, 2010). Poverty is widespread, increasing vulnerability to these effects (Shi et al., 2016), and the COVID-19 pandemic is likely to severely affect progress on sustainable development and biodiversity conservation.

This exceptionally high vulnerability to climate change has led to a focus on climate adaptation, but the government of Bangladesh has also committed to a greenhouse gas (GHG) reduction of 21.85% below business-as-usual by 2030, of which 15% is conditional on international support (MoEFCC, 2021). Bangladesh also has an ambition to become an upper middle-income country over the next decade, and the updated Nationally Determined Contribution (NDC) states that the GHG goals should not undermine the national principles of maintaining minimum 8% GDP growth, eradicating poverty by 2030, and ensuring food and nutrition security for all citizens. Bangladesh’s NDC further aims at a long-term vision for synergies between adaptation and mitigation actions (MoEFCC, 2021).

Bangladesh has several key natural assets, including two thirds of the Sundarbans (the largest remaining area of mangroves in the world), the Chittagong hill forests in the east, and the unique seasonal wetlands (Haors) in the north-east, but 60% of the country is cropland (FAOSTAT, 2018). Both natural and managed ecosystems are being degraded due to climate change, pollution and over-exploitation of resources, posing an increasing threat to livelihoods, especially for the rural poor (Rasul and Thapa, 2004; Miah et al., 2010; Abdullah-Al-Mamun et al., 2017). NbS offer the potential to reverse this degradation and boost climate resilience, whilst empowering local communities and enabling sustainable development, but they are not well integrated into national policies (Islam et al., 2021), partly due to lack of awareness of their benefits (Huq et al., 2017).

This review aims to 1) identify robust evidence on the effectiveness of NbS in Bangladesh for addressing climate change, natural hazards and other sustainable development goals; and 2) assess the enabling factors that can accelerate and expand the uptake of good quality NbS. We build on the methodology of two recent assessments: a global systematic map of evidence on the effectiveness of nature-based interventions for adapting to the impacts of climate change (Chausson et al., 2020) and a review of the outcomes of NbS on development in lower-income countries (Roe et al., 2021). We expanded the scope of these global reviews to carry out a deeper analysis and synthesis for one country. This evidence highlights the benefits provided by NbS, and their potential to help developing countries reach their economic and environmental goals.

METHODS

Systematic Review Protocol

Target Interventions

NbS are defined as actions to protect, sustainably manage, and restore natural or modified ecosystems, that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits (Cohen-Shacham et al., 2019; Seddon et al., 2021). For example, protecting and restoring forests can help to reduce the impacts of floods and landslides, while nature-based agricultural techniques such as the use of leguminous cover crops can improve the water-holding capacity of the soil, helping to combat droughts. We include modelling studies that assess the potential benefits of NbS that have not yet been implemented, and, similarly, we include assessments of the benefits delivered by existing ecosystems, because this evidence is a useful proxy for the benefits that would be delivered through protecting, restoring, or managing the ecosystems through future NbS actions. We use this approach because our aim is to gather evidence on the potential future effectiveness of scaling up the deployment of NbS in Bangladesh, not only on the benefits currently being delivered by NbS that are already implemented.

NbS should be designed and implemented with the full engagement and consent of Indigenous Peoples and local communities, and must sustainably provide one or more benefits for people whilst causing no loss of biodiversity or ecological integrity (or preferably a gain) compared to the pre-intervention state (Seddon et al., 2021). It was rarely possible to determine whether all these criteria were met based on the information given in the papers, hence some interventions may not qualify as 'solutions', as per the NbSI guidelines and the IUCN standard (NbSI, 2020; IUCN, 2020). However, we recorded any relevant information on biodiversity and social impacts, even where it seemed possible that mixed or negative impacts might have occurred, because this is important in highlighting lessons for NbS design in future. We considered the likely baseline or counterfactual scenario, i.e. what would have happened in the absence of the intervention. We therefore included interventions such as agro-ecological farming methods if the most likely alternative was a continuation of more damaging practices, and sustainable fishery management if the alternative was a continuation of over-harvesting. We use the term biodiversity in its broadest sense, with a positive outcome for biodiversity indicating a move towards an appropriate mix and abundance of habitats and species for each location, acknowledging that some ecologically valuable habitats have naturally low species diversity.

Target Outcomes

The strength of NbS is that they can simultaneously address multiple challenges. Outcomes can arise either from changes to ecosystems to support or increase the provision of ecosystem services or through the process of implementing the NbS, such as through training, employment, and empowerment. In this way, NbS can address climate and natural hazards at the same time as contributing to other sustainable development goals. They can reduce vulnerability to climate change and natural hazards by reducing exposure to impacts (e.g., forests protecting against floods), reducing sensitivity to impacts (e.g., by diversifying livelihood options) and increasing the capacity to adapt to change (e.g., by empowering communities and individuals) (Thiault et al., 2021). In addition, GHG mitigation reduces hazards by limiting the magnitude of climate impacts, and biodiversity underpins the adaptive capacity and healthy functioning of the ecosystem, for example by maintaining genetic diversity that could confer resilience to future pests and disease. We drew up a list of relevant NbS outcomes adapted from Roe et al. (2021), and identified how they address climate and natural hazards and contribute to the SDGs (**Table 1**). These are hereafter referred to as the 'target outcomes' for the review. Further details are provided in the **Supplementary Information**.

The systematic review was based on the methodology of Chausson et al. (2020), which used the scoping elements listed in the top row of **Table 2**. In order to restrict their global search to a manageable number of articles, Chausson et al. (2020) searched academic articles only, and excluded the categories shown in the middle row of **Table 2**. These criteria retrieved only two papers for Bangladesh. For this in-depth country-level study we removed all these exclusion criteria, as shown in the bottom row of **Table 2**, and we also searched for evidence from books, conference proceedings and grey literature.

Search and Screening Process

Academic Literature

The search string (see **Supplementary Information**) included recognized intervention terms (e.g., nature-based solution, ecosystem-based adaptation, agroforestry), the people or sector benefiting from NbS (e.g., local communities, policymakers, food systems), the challenge targeted (e.g., climate change, flood, drought, landslide) and the outcome (e.g., food security, adaptation, mitigation, protection, resilience). The string was based on that used by Chausson et al. (2020) but after review by the co-authors based in Bangladesh we included one additional local term ("floating gardens"). We also added an extra step in which we simply searched for "Nature-based solutions" (and related terms) and "Bangladesh". This was to check that we had not inadvertently screened out any relevant studies due to the complex and specific search terms used for the main search. We searched only for studies of NbS in Bangladesh.

TABLE 1 | Target outcomes from NbS for addressing climate change and other SDGs, either through ecosystem change or through the NbS implementation process.

Broad category of outcome	Outcomes from the NbS	Addresses				Through	
		Exposure	Sensitivity	Adaptive capacity	SDGs[a]	Ecosystem	NbS implementation
Climate change mitigation	GHG concentrations	x			13	x	
Inland flooding and erosion	Inland flooding	x			13	x	
	Soil erosion	x			2, 13	x	
	Mudslides/landslides/avalanche	x			13	x	
Coastal flooding, erosion and salinization	Coastal flooding	x			13	x	
	Coastal erosion	x			13	x	
	Coastal saltwater intrusion (groundwater)	x			13	x	
	Salinization (surface)	x			2, 13	x	
Wind damage	Wind and storm damage (other than flooding)	x			13	x	
Heatwaves	Heatwaves	x			3, 11, 13	x	
Wildfire	Wildfire	x			13	x	
Desertification	Desertification	x			13	x	
Water security	Drought/reduced rainfall	x			2, 6, 13	x	
	Water quantity/availability	x			6	x	
	Surface water quality	x			6	x	
	Groundwater quality	x			6	x	
Food security	Food production/security/nutrition		x		1, 2	x	
	Fishing		x		1, 2	x	
	Aquaculture		x		1, 2, 12	x	
	Soil quality		x		2	x	
	Pests		x		2	x	
Wood, fuel and NTFP (Non-timber forest products)	Wood production (forestry)		x		1, 12	x	
	Fuelwood supply		x		1	x	
	Biofuel production		x		1	x	
	Other ecosystem goods (e.g. NTFP)		x		1, 12	x	
Air quality	Air quality	x	x		3	x	
Disease risk	Disease incidence and distribution	x	x		3	x	
Cultural outcomes	Aesthetic value		x		3	x	
	Recreation (local)		x		3	x	
	Cultural heritage, spiritual values and inspiration		x	x	3	x	x
Socio-economic outcomes	Tourism		x		1, 3, 8	x	x
	Employment		x		1, 8	x	x
	Local economic benefits		x		1, 8	x	x
	Education and training		x	x	1, 4	x	x
	Rights, empowerment and inequality (incl. gender)		x	x	5, 10	x	x
	Social cohesion, governance and engagement		x	x	16, 17		x
Ecological outcomes	Biodiversity and ecosystem health			x	14, 15	x	

[a] *SDGs: 1 No poverty, 2 Zero hunger, 3 Good health and well-being, 4 Quality education, 5 Gender equality, 6 Clean water and sanitation, 7 Affordable and clean energy, 8 Decent work and economic growth, 9 Industry, innovation and infrastructure, 10 Reduced inequalities, 11 Sustainable cities and communities, 12 Responsible consumption and production, 13 Climate action, 14 Life under water, 15 Life on land, 16 Peace, justice and strong institutions, 17 Partnership for the goals.*

We searched Web of Science and Scopus on May 7, 2020 for articles, reviews, conference proceedings, reports or book chapters that matched these terms. We excluded duplicates using EndNote, and screened first titles and then abstracts to eliminate sources that clearly did not contain any evidence of NbS effectiveness for delivering the target outcomes. As the aim of this review was to conduct an in-depth analysis of the strongest evidence, we then performed a further screening round, selecting only the sources that explicitly referred to evidence on the effectiveness of NbS for delivering the target outcomes in the abstract. The sources excluded at this stage included many general texts or reviews about climate change adaptation or mitigation, some of which did not explicitly refer to NbS in the abstract, most of which appeared to consist largely of secondary information taken from other studies. However, it is possible that some of these sources may contain some primary evidence on NbS effectiveness. Finally, some additional studies were excluded at the full-text screening stage, either because they were duplicate studies or because they were not relevant.

Grey Literature

Many NbS projects in Bangladesh are not included in peer-reviewed journals. It was therefore important to analyze grey literature on

TABLE 2 | Framing of the search criteria for the systematic review: differences to the global review by Chausson, Turner et al. (2020).

Subject/Population	Intervention	Comparators	Outcomes
Chausson, Turner et al. (2020) criteria			
Human individuals, groups, communities and economic sectors (e.g., agriculture, water, forestry, transport, energy)	Actions in rural, semi-rural or peri-urban settings involving management, restoration or protection of biodiversity, ecosystems, or ecosystem services, or involving the creation or management of artificial ecosystems (excluding agriculture, fisheries and aquaculture)	Pre-intervention baselines or repeat assessments over time; quasi or experimental controls (no adaptation action); modeled counterfactuals, or evaluator inference of a counterfactual (i.e. what would have happened in the absence of the intervention)	Measured, observed, or ex-ante modeled outcomes (regulating or provisioning ecosystem services) addressing the impacts of weather hazards or climate change on people or economic sectors
Exclusions in Chausson, Turner et al. (2020)			
1. Effects of nature-based interventions on impacts not explicitly reported as being driven (at least in part) by climate or hydro-meteorological phenomena 2. Effects on vulnerability (including social adaptive capacity) only arising from the implementation, management or governance of the nature-based intervention, rather than (at least in part) from the flow of ecosystem services 3. Urban nature-based solutions, hybrid natural/engineered interventions, agricultural interventions (such as agroforestry), rangeland, or fisheries interventions not involving ecosystem restoration or protection 4. Effectiveness of existing ecosystems for adaptation relevant services, unless an intervention (e.g. protection or restoration) was involved			
Modifications to criteria for Bangladesh review			
Same	Urban NbS, hybrid NbS, agriculture, fisheries and aquaculture included Effectiveness of existing ecosystems was included even in the absence of an explicit intervention	Same	Included all the outcomes in **Table 1**

these interventions. Because non-academic search engines return very large numbers of hits, most of which are not relevant, we used the knowledge of local experts and networks to narrow down the search to the most relevant resources. From our personal knowledge and experience of the implementation of NbS in Bangladesh, together with discussion with members of the 'NbS Bangladesh Network', a community of researchers, practitioners and policymakers (www.nbsbangladesh.info), we identified four major projects implemented in Bangladesh over the past 22 years. These projects focused on community-based natural resource management, ecosystem-based adaptation and biodiversity conservation, and covered multiple sites in the Sundarbans, the coast, the Chittagong Hills and the Haor wetlands. We searched for reports from these projects through online sources and personal contacts, and finally selected three final reports (one in two volumes) (DoE, 2015; IUCN Bangladesh, 2016, MACH-II, 2007a; MACH-II, 2007b) and one performance report (Winrock International, 2018). From these we identified 24 interventions that qualify as NbS.

Many other projects followed ecosystem-based approaches in Bangladesh, but documents with adequate evidence were not available. This reflects challenges with the grey literature evidence base. Documents often state outcomes (mainly in qualitative terms) without fully describing the methodology used to determine the outcome, so it is not possible to assess the robustness of the information, and often they do not follow a consistent impact assessment methodology throughout the whole project period.

Coding Strategy

For each paper or report reviewed, we extracted data on the NbS interventions and their outcomes into a spreadsheet, based on a coding template adapted from Roe et al. (2021) (see **Supplementary Information**). For each NbS intervention we recorded the location, NbS type (protection, restoration, management or creation of ecosystems, or nature-based food production), ecosystems involved, funders, instigators, partners, beneficiaries, economic costs of implementation, and synergies and trade-offs between outcomes. We also collected information on 'enabling factors' reported to influence the successful implementation and governance of the NbS, including the role of institutions, the involvement of local communities, and the use of local knowledge.

For each outcome of an intervention, we recorded the type of outcome (**Table 1**), direction of outcome (positive, negative, mixed, unclear), the attributes of the ecosystem that influenced the outcome (e.g., species richness; presence of particular species); the methods used to determine the outcome, and the quality of the evidence. All financial amounts are presented as they appeared in the corresponding literature. In 2021, US$ 1 was equivalent to approximately Bangladesh Taka 85.

We assessed the quality of the evidence using the protocol in the **Supplementary Information (Section S3.17)**. We recorded whether there was any conflict of interest declared, or if the authors were also involved in implementation of the study, although these papers were not excluded. For each outcome reported, we then recorded whether primary evidence was used and displayed, or secondary evidence provided with references; whether the methodology was clear and appropriate; whether results were reported with respect to a counterfactual or baseline (if appropriate); and whether confounding factors were taken into account. Outcomes that met all these criteria were deemed to have robust evidence.

We also recorded any examples mentioned in each study of 'enabling factors' that enabled the successful deployment of NbS.

Analysis and Synthesis of Results

We produced descriptive statistics of the results, including the number of interventions studied in different ecosystems, and the number of positive, negative, mixed or unclear outcomes for each type of intervention (*Number of Studies and Quality of the Evidence Base* and *Type of Nature-based Solutions Interventions*). We then synthesized evidence on the effectiveness of NbS interventions for addressing climate change, natural hazards and other sustainable development goals, described in narrative form with supporting examples in a table (*Effectiveness of Nature-based Solutions for Addressing the Target Outcomes*). Finally, we synthesized information on enabling factors for scaling up high quality NbS and presented this in narrative form (*Enabling Factors for Successful Implementation of Nature-based Solutions*).

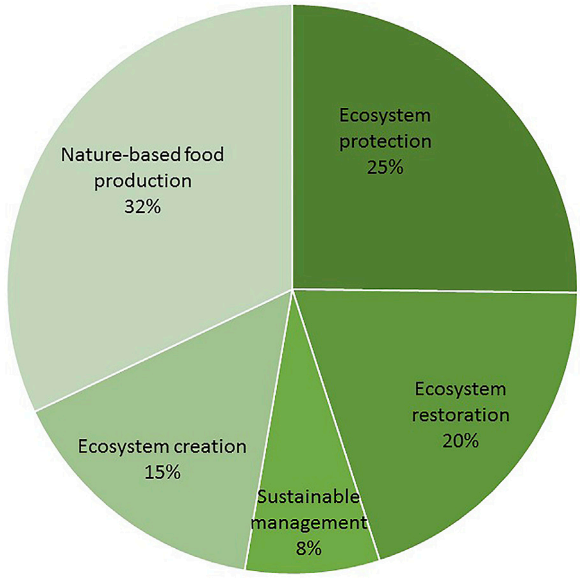

FIGURE 1 | Type of NbS intervention included in the studies that were reviewed.

RESULTS

Number of Studies and Quality of the Evidence Base

The search of academic literature retrieved 1,173 non-duplicate articles of which 56 remained for coding after all the screening stages (**Supplementary Figure S1**, **Supplementary Table S1**). Five papers that contained no outcome evidence did contain useful material on enabling factors; These were coded in the spreadsheet.

These 56 papers reported 154 outcomes, of which 96 (62%) had robust evidence, i.e., they reported and displayed primary data, they used a clear and appropriate methodology, used a counterfactual or baseline (if applicable), and they attempted to account for confounding factors (if applicable). Of the outcomes, 115 were based on quantitative data (75%), and 46 (30%) reported qualitative data, with seven of these (5%) using both. Sixteen of the outcomes were based on experiments, 14 on modelling, 77 on interviews and 34 on *in-situ* observations, of which 16 also used interviews. Six outcomes were based on literature reviews, and one was based only on anecdotal evidence. For 88 outcomes across 21 academic studies, participatory approaches were used, including through interviews and focus groups. This included approaches incorporating traditional or indigenous knowledge, including knowledge of local farming techniques, crop cultivars, crop pests, cultural values and use of forest products.

None of the 85 outcomes reported in the grey literature were based on robust evidence. Also, as the authors were involved in the interventions, the reports cannot be considered as independent evaluations. Only 36% of the outcomes were reported as quantitative evidence (of which 21% also provided qualitative evidence), and this often concerned intermediate outputs (e.g., the area of habitats restored, the number of community organizations established or the number of training courses provided) rather than final outcomes. Most of the outcomes (64%) were based only on qualitative evidence, sometimes from interviews and surveys of community perceptions of benefits, but often it was not clear how the evidence was obtained. Biophysical outcomes such as flood and erosion prevention were often reported as inferred or expected outcomes resulting from an intervention (e.g., 'vegetation was planted to provide protection from flooding and erosion'), and thus could not be included in the evidence base. The lack of quantitative data also limited the scope for economic analysis of costs or benefits. However, the reports often provided rich detail of real-life governance, engagement and capacity building challenges and lessons on how these could be addressed.

Type of Nature-based Solutions Interventions

The most frequent type of intervention in the literature reviewed was nature-based food production (32%), followed by protection and then restoration of ecosystems (**Figure 1**). Although only 8% of interventions are classified as solely 'ecosystem management', many of the nature-based food production studies also involved management (e.g., of cropland or fisheries).

The most common ecosystem involved in the NbS interventions was cropland, followed by inland wetlands, agroforestry, mangroves, tropical forests, and plantations, with just a few studies of NbS in other habitats (**Figure 2**). There was a notable gap for urban green and blue infrastructure, with only one study, focusing mainly on the role of roof gardens for food production (Zinia and McShane, 2018). From the combination of the type of intervention (protection, restoration etc.), the ecosystem involved, and terms used to describe certain types of intervention (e.g., "conservation agriculture") we generated a

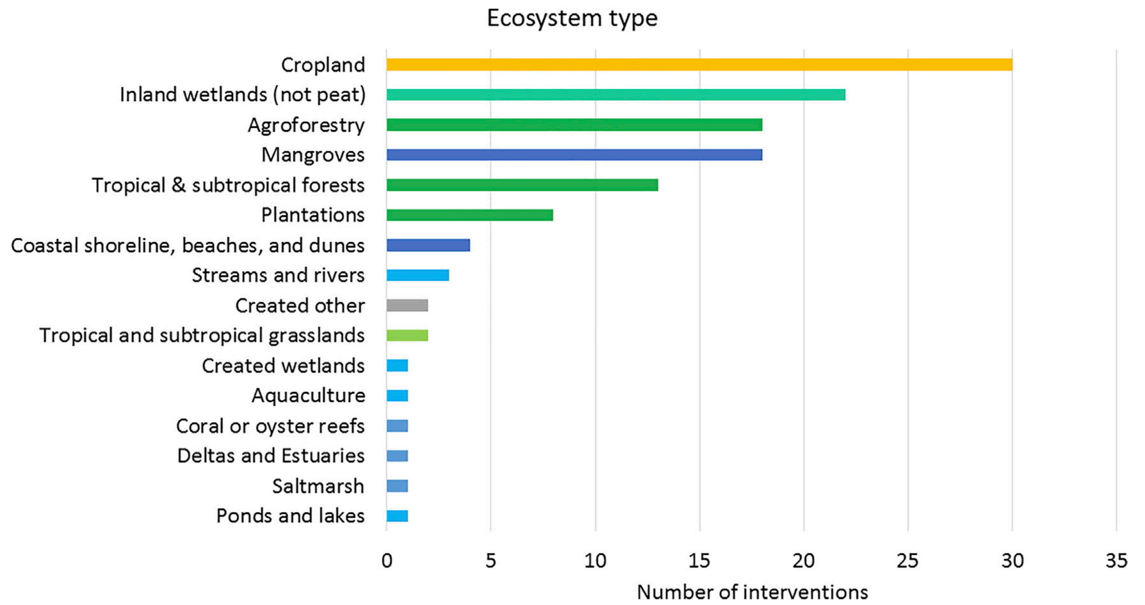

FIGURE 2 | Number of interventions reviewed for each ecosystem type.

list of the main types of interventions identified in the review (**Table 3**).

Effectiveness of Nature-based Solutions for Addressing the Target Outcomes

The review identified evidence on the capacity of NbS to address all the target outcomes except wildfires, desertification, and the spread of diseases, for which no evidence was found. The most frequently recorded outcomes were for food production, climate change mitigation, biodiversity, fishing, and coastal flooding (**Figure 3**). Most (91%) were positive, with only 3% being negative and 2% mixed (the rest were neutral or unclear).

We summarized the positive links between different NbS interventions (**Table 2**) and target outcomes (**Table 1**) in **Table 4**. The most frequently reported evidence was for the coastal protection and socio-economic benefits of mangroves (15 positive outcomes each), followed by the benefits of conservation agriculture for food security (13 positive outcomes). These were also the most frequently reported outcomes when looking only at robust evidence (**Supplementary Table S6**).

From the information reported in the studies, we recorded which attributes of the ecosystem positively influenced the outcomes. Out of the 228 outcomes reported, 96 depended on the presence of a specific habitat or ecosystem, 61 were influenced by species abundance, 57 required the presence of a specific functional group such as trees, fish or birds, 44 were influenced by species richness or diversity (including 24 biodiversity outcomes), 18 by soil carbon or soil health and one (slope stabilization) by root morphology (Islam and Rahman, 2019), with some overlaps, i.e., some outcomes were influenced by more than one ecosystem attribute.

In the following sections we synthesize the evidence on the effectiveness of NbS for addressing each of the target outcome groups in Bangladesh, and show how these are linked to the Sustainable Development Goals. For clarity, details and citations for many of the examples used in this section are presented in **Table 5**.

Greenhouse Gas Reduction

There were 22 records of positive outcomes for GHG reduction and one mixed outcome. Sixteen were from forests and six from conservation agriculture.

Protecting and restoring mangrove forests is particularly important as they trap carbon-rich sediment amongst their roots, as well as storing carbon in biomass. Carbon storage was estimated at 219 tC/ha (Rahman et al., 2017) to 257 tC/ha, of which 63% was belowground in the soil and roots (Abdullah-Al-Mamun et al., 2017). However, the global average for oceanic mangroves was estimated as 400 tC/ha, suggesting that mangroves in Bangladesh could be relatively degraded (Chow, 2018). Mangroves were estimated to sequester carbon four times faster than mature land-based forests, offsetting 1.5% of Bangladesh's fossil fuel carbon emissions in 2014 or 10% from 1997 according to different estimates (**Table 5**). Conversion to aquaculture was highlighted as a threat, as it requires excavating at least 2 m of sediment, which can release 70 tC/ha of carbon (Chow, 2018).

We found fewer studies on carbon storage and sequestration in **native terrestrial forests**, which include the mixed evergreen forests in the Chittagong Hill Tracts and the much smaller fragments of deciduous Sal forests (*Shorea robusta*) in central Bangladesh. Progress towards protecting and restoring forests for carbon benefits via REDD+ faces governance challenges in Bangladesh, especially in the Chittagong Hill Tracts (Richards

TABLE 3 | Main types of NbS intervention identified in the review for Bangladesh.

Ecosystem	Type of intervention	No	Description
Coastal	Mangrove protection and restoration	15	Protection of existing or restored mangroves or replanting of mangrove seedlings for coastal protection, livelihoods and biodiversity. Most (9) studies are in the Sundarbans reserve, an area of over 600,000 ha which has been protected as a Ramsar site since 1992, but some (6) assess restoration initiatives in the Chittagong region or along the south coast
	Oyster reef creation	1	One experimental site on the south coast, to assess the benefits for coastal protection
	Shoreline conservation	1	Mixed conservation measures in Cox's Bazar - Teknaf Peninsula and Sonadia Island Ecologically Critical Areas (ECAs) including protecting mudflats and the rocky intertidal zone and conservation of sea turtles, to benefit fisheries
	Sand dune revegetation	2	Replanting native vegetation in Cox's Bazar - Teknaf Peninsula and Sonadia Island ECAs for erosion protection
	Coastal shelterbelt	2	Planting of strips of coastal trees to protect from storm surges, cyclones and coastal erosion
Inland wetlands	Swamp forest protection and restoration	4	Planting native swamp forest trees in the Haor wetlands for flood and erosion protection, livelihoods and biodiversity
	Wetland protection and restoration	8	Protecting and restoring the Haor wetlands, e.g. re-excavating silted up areas that dry out in summer, to protect fisheries
	Fishery management	4	Regulating fishing and preventing 'poison fishing' to avoid over-exploitation, in the Haors
	Floating gardens	2	Growing vegetables on mats of floating wetland vegetation (mainly water hyacinth) when farmland is inundated
	Bioremediation	4	Use of water hyacinth to remediate water pollution in the Haors; and experiments with constructed wetlands or soil fungi to remove arsenic from water and soils
Terrestrial forests, shrub and grass	Terrestrial forest protection and restoration	9	Protection and restoration of the Chittagong Hill Tracts forests, to protect from erosion and sustain livelihoods. Also planting trees to stabilize embankments in the Haors
	Community forestry	4	Vulnerable local people co-manage plantations or native forests and in return are allowed to harvest them sustainably
	Forest plantation	1	Experiment to assess carbon storage and sequestration
	Grass and shrub cover	2	Use of native grasses and shrubs on embankments and around homesteads to protect against erosion
Agroforestry and homegardens	Agroforestry	6	Planting rows of fruit, timber or fuelwood trees amongst other crops; helps to stabilize soil on steep slopes
	Homegardens	7	Small areas around homesteads, growing a diverse mix of trees, shrubs, vegetables and other plants for food, timber, fuel, ornamental and medicinal use
Cropland	Conservation agriculture	15	Experiments or large scale field trials of conservation agriculture techniques including reduced tillage, retaining crop residue, adding organic matter (e.g., manure, compost) to soils, cultivating rice without flooding the field, increasing crop diversity, and integrated pest management. Aimed at increasing resilience to climate change (especially droughts) and reducing the use (and cost) of agro-chemicals
	Rainwater harvesting	1	Excavating ponds to store water for use during the dry season
Urban	Urban green space	1	Production of food and other goods in rooftop gardens and other open spaces in Dhaka

and Hussain, 2019). Low income households are highly dependent on forests for fuelwood and timber, and over-harvesting is causing severe forest degradation which reduces carbon storage (Yong Shin et al., 2007). For example, one study found that plots reforested with species that were valued as wood fuel showed much lower levels of carbon storage than those using other species, due to harvesting by local people. It was suggested that providing secure land tenure and clean energy options for indigenous communities could enable more sustainable management of forest resources with benefits for carbon storage (Yong Shin et al., 2007).

Forests managed for production are only categorized as nature-based solutions if they provide biodiversity benefits (compared to business-as-usual) and are co-implemented with local communities. Carbon storage will be offset by emissions from wood extracted for fuel or short-lived products such as paper. Estimates of carbon storage are lower than for natural forests, ranging from 200 tC/ha for roadside social forestry plantations (Rahman et al., 2015) to 118 tC/ha for homegardens (Nath et al., 2015) and 31–37 tC/ha for agroforestry (Hanif et al., 2015). However homegardens in Bangladesh stored more carbon than those in India, which could be due to their higher tree density and diversity (Nath et al., 2015).

Conservation agriculture can play an important part in reducing emissions by improving soil health and soil carbon. This enables inputs of mineral fertilizers to be reduced and thus also cuts nitrous oxide emissions (Islam et al., 2011; Alam et al., 2020), and it may reduce the need for pumped irrigation, saving carbon emissions from fuel use (Begum et al., 2018). However, switching to 100% organic fertilizers may not be the best strategy as although it increases soil carbon, it may reduce yields and increase methane emissions (Begum et al., 2018). Also, cow dung and crop residues are in demand as fuels

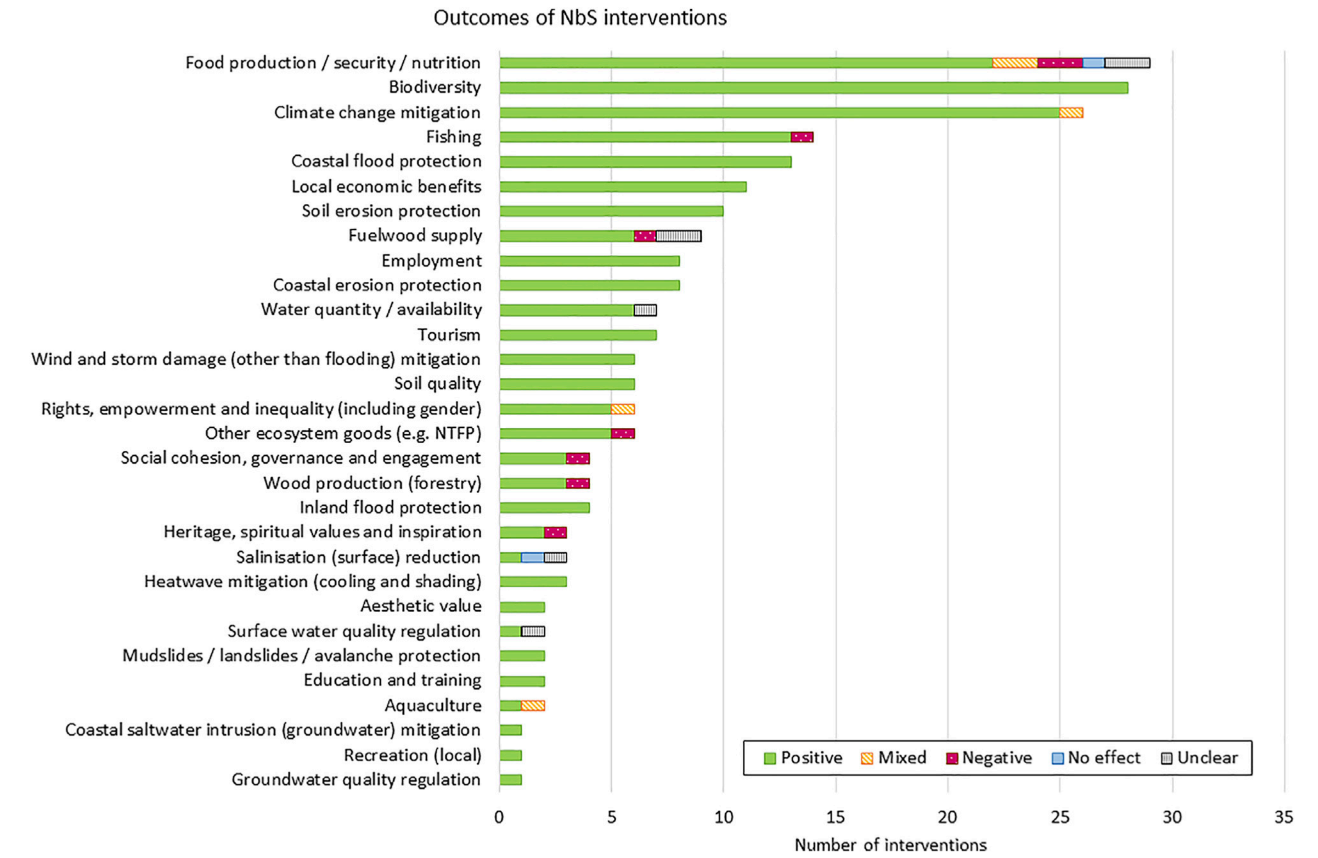

FIGURE 3 | Number of interventions with positive, negative or mixed outcomes for addressing climate and natural hazards and other sustainable development goals.

(Begum et al., 2018). Integrated methods that combine organic fertilizers with reduced tillage, increased residue retention and lower use of synthetic fertilizers can successfully maintain yields whilst cutting costs and emissions (Islam et al., 2011; Begum et al., 2018).

Non-puddled transplanting of rice is an important option for reducing methane emissions. It involves strip tillage (tilling only the strips to be planted rather than the whole field) followed by planting into saturated soil, rather than the usual practice of planting into a ploughed and flooded field (Bell et al., 2019). This cuts the total lifecycle greenhouse gas emissions of rice production by between 16 and 31% depending on the level of crop residue (straw) retention (Bell et al., 2019). Residue increases soil carbon storage and crop yield, but this is offset by higher methane emissions as the incorporation of residue into the soil stimulates the activity of methane-producing bacteria (Alam et al., 2019).

Coastal Floods, Erosion, and Salinization

Bangladesh is one of the most vulnerable countries in the world to coastal hazards (Shi et al., 2016). A large proportion of the population live in low-lying coastal areas on a funnel-shaped delta with a high tidal range, putting them at high risk from cyclones and storm surges (Das et al., 2010; Rahman et al., 2019) which cause extensive wind and flood damage. In addition, coastal areas are often lost to erosion, and suffer from salinization of groundwater and agricultural land due to seawater intrusion and frequent flooding. We found 24 outcomes for coastal protection: 22 positive, 1 mixed and 1 unclear. Most (15) were for mangrove protection and restoration.

Mangroves have provided a natural barrier to coastal hazards for centuries. They are estimated to protect 1.1 million to 3.5 million people in Bangladesh from coastal flooding during cyclones (Akber et al., 2018), avoiding damage worth at least US$1.56 billion per year on average (Menéndez et al., 2020). Villages protected by mangroves had only about half of the monetary loss from flood and wind damage associated with cyclone Sidr, compared to other villages (Akber et al., 2018). Even a 100 m deep coastal shelterbelt of healthy mangroves can reduce storm surge velocity by up to 92%, protecting embankments from costly damage (Dasgupta et al., 2019), and a double shelterbelt of mangrove and *Casuarina* trees, 200–300 m in depth, can reduce storm surge height by up to 22% and velocity by up to 49% (Das et al., 2010).

The Forest Department of Bangladesh started planting mangroves along the whole coastline in 1966, initially aiming to boost protection against cyclones and storm surges, and later to stabilize newly accreted (char) land so that it can be used for agriculture (Iftekhar and Takama, 2008). Planting with mangroves greatly increased the ratio of accretion to erosion,

TABLE 4 | Number of positive outcomes reported in the literature reviewed for each type of NbS.

		GHG reduction	Coastal floods, erosion and salinization	Inland floods and erosion	Wind damage	Heatwaves	Water security	Food security	Wood, fuel and NTFP	Cultural benefits	Socio-economic benefits	Biodiversity	Total
Coastal	Mangrove protection and restoration	7	15	2	3	1	0	4	6	3	15	4	32
	Oyster reef creation	0	1	0	0	0	0	0	0	0	0	0	0
	Shoreline conservation	0	0	0	0	0	0	1	0	0	0	1	2
	Sand dune revegetation	0	3	0	0	0	0	0	0	0	0	1	1
	Coastal shelterbelt	1	2	0	1	0	0	0	1	1	1	1	4
Inland wetlands	Swamp forest protection and restoration	1	0	2	0	0	0	3	0	0	4	4	11
	Wetland protection and restoration	0	0	0	0	0	0	5	0	0	9	6	20
	Fishery management	0	0	0	0	0	1	3	0	1	3	4	11
	Floating gardens	0	0	0	0	0	0	2	0	0	1	0	3
	Bioremediation	0	0	0	0	0	2	1	0	0	0	0	1
Terrestrial forests, shrubs and grass	Terrestrial forest protection and restoration	2	0	7	1	1	0	2	0	0	5	3	10
	Community forestry	1	1	0	0	0	0	1	0	0	2	1	4
	Forest plantation	1	0	0	0	0	0	0	0	0	0	0	0
	Grass and shrub cover	0	0	2	0	0	0	0	0	0	0	0	0
Agroforestry and homegardens	Agroforestry	2	0	1	1	0	0	4	3	0	3	2	12
	Homegardens	1	0	2	0	1	0	1	2	0	4	1	8
Cropland	Conservation agriculture	6	0	0	0	0	4	13	0	0	3	0	16
	Rainwater harvesting	0	0	0	0	0	1	0	0	0	0	0	0
Urban	Urban green space	0	0	0	0	0	0	3	2	0	0	0	5
Total		22	22	16	6	3	8	43	14	5	50	28	140

TABLE 5 | Key examples of evidence on the target outcomes of NbS cited in *Effectiveness of Nature-based Solutions for Addressing the Target Outcomes*

Outcome	NbS	Selected examples of outcome evidence	Reference
GHG reduction	Protecting and restoring mangroves	Carbon storage: 219 tC/ha	Rahman et al. (2017)
		Carbon storage: 257 tC/ha, of which 63% below ground in the soil and roots	Abdullah-Al-Mamun et al. (2017)
		Carbon storage (global averages): 400 tC/ha for oceanic mangroves; 2000 tC/ha for estuaries. Average for Indo-Pacific region: 1,023 tC/ha. Conversion to aquaculture by excavating >2 m of sediment can release 70 tC/ha. Sequestration: 1.5 to 6 tC/ha/y (global range)	Global review. Chow, (2018)
		Sequestration: 1.7 tC/ha/y, four times more than mature land-based forests. Offset 1.5% of Bangladesh's fossil fuel CO_2 emissions in 2014	Global review. Taillardat et al. (2018)
		Sequestration: Sundarbans sequestered 4.8 Mt CO_2/year from 1997 to 2010. Offset 10% of Bangladesh's CO_2 emissions	Abdullah-Al-Mamun et al. (2017)
	Plantations	Carbon storage: Roadside social forestry plantations in south-western Bangladesh store almost 200 tC/ha although this is less than native woodlands	Rahman et al. (2015)
	Agroforestry	Sequestration: 115–135 tCO2/ha/y (equivalent to 31–37 tC/ha/y) 7 years after planting for three typical fast-growing species	Hanif et al. (2015)
	Homegardens	Carbon storage: average 118 tC/ha in above-ground biomass, much higher than homegardens in India, thought to be due to higher tree density	Nath et al. (2015)
		Carbon storage: Soil organic carbon 0.12–1.65%; positively correlated with tree species diversity and density, probably because more diverse systems are more productive due to niche complementarity	Islam et al. (2015)
Coastal floods, erosion and salinization	Protecting and restoring mangroves	820 km^2 and 1.1 million people protected from coastal flooding during tropical cyclones and other storms by mangroves in Bangladesh, avoiding damage worth US$1.56 billion per year on average	Global model Menéndez et al. (2020)
		3.5 million people protected by mangroves in Sundarbans Villages protected by mangroves had about half of the monetary loss from flood and wind damage to houses, property, crops, livestock and aquaculture stock associated with cyclone Sidr (TK 69,726, US$1,025 per household), compared to villages not protected by mangroves	Survey of Sundarbans Akber et al. (2018)
		Even a 100 m deep strip of healthy mangroves can reduce storm surge velocity for a storm of the same magnitude as cyclone Sidr by up to 92%, providing significant savings in maintenance costs by protecting embankments from damage	Model Dasgupta et al. (2019)
		Char land areas in the Barisal and Chittagong regions planted with mangroves experienced 37.2 times more accretion than erosion between 1973 and 1989 and 4.7 times more from 1989 to 2010, compared to only 1.6 and 1.3 times more accretion than erosion in areas that were not planted. For lands that were newly accreted in 1989, 31% of non-plantation land had eroded by 2010 compared to only 10% of plantation land	GIS analysis Chow, (2018)
		The CBA-ECA project planted 361 ha of mangrove and 62 ha of sand dune vegetation to protect the flora and fauna of the Cox's Bazar-Teknaf Peninsula and Sonadia Island ECAs. Community-based organizations worked with local and national government and law enforcement agencies to revert illegal shrimp farms into mangrove forest	Long term project report DoE, (2015)
		The CREL project supported planting of 565,000 mangrove seedlings on 512 ha, estimated to deliver US$485 million of storm protection services as well as co-benefits totaling US $684 million annually, and planted 20 ha of sand dunes with 562,000 seedlings of Nishinda and Dholkolmi (*Ipomoea carnea*) to reduce erosion and storm surge impacts, helping to maintain the integrity of the island, and creating habitats to support indigenous species	Long term project report (Winrock International, 2018)
	Coastal shelterbelts	Double coastal shelterbelts of mangrove and *Casuarina* trees, 200–300 m in depth, can reduce storm surge height by up to 22% (1.4 m) and surge velocity by up to 49% (1.2 m/s) for an event like Cyclone Sidr. Mangroves have a higher drag due to their aerial root structure, but densely planted *Casuarina* can be more effective for very high surges Das et al. (2010)	Model Das et al. (2010)
Inland flooding and erosion	Agroforestry	Cultivation of cash crops such as ginger causes soil erosion that costs about 11% of the total production costs, but agroforestry can turn this loss into a gain of about US$26 ha/year, as the soil-formation rate exceeds the erosion rate	Rasul, (2009)
		In the MACH project (Table S5), switching from planting pineapples along contours rather than in rows up and down slopes also allowed denser planting and resulted in increased fruit size, thus increasing the farmers' income and food security. The contour plots increased profits by 140% over 3 years, to Tk 128,600 per acre, which is Tk 74,990 per acre more than the traditional cultivation system	Long term project report MACH-II, (2007a), MACH-II, (2007b)

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TABLE 5 | (*Continued*) Key examples of evidence on the target outcomes of NbS cited in *Effectiveness of Nature-based Solutions for Addressing the Target Outcomes*

Outcome	NbS	Selected examples of outcome evidence	Reference
	Protecting and restoring terrestrial forests	Catchments with regenerating or planted trees and other vegetation had 3–4 times less soil erosion, 4–35 times less nutrient loss, 16% less annual runoff and the peak flow was seven times lower than a catchment that had been cleared for agriculture	Paired catchment experiments Gafur et al. (2003)
		Local tree species with deep tap roots could successfully stabilize steep slopes at risk of landslides although this is only suitable for slopes of less than 70°	Model Islam and Rahman, (2019)
		In the Haor basin, Government agencies, donors, NGOs and local communities collaborate to plant locally raised seedlings of flood-tolerant plants, particularly Hijal (*Barringtonia acutangula*) and Koroch (*Pongamia pinnata*). 125,000 seedlings were planted onto kanda (raised land) under the Tanguar Haor Project (2006–2016), 18,450 seedlings were planted in 17 ha at six sites at Hakaluki Haor under the CBA-ECA project (2011–2015), and 112 ha were restored under the CREL project In Hakaluki Haor, 11 submersible embankments with 'green belts' of indigenous tree species were established	Long term project reports DoE, (2015), IUCN Bangladesh, (2016), Winrock International (2018)
	Growing herbaceous plants	Villagers in Chanda Beel are growing low plants such as *Sesbania* (doincha), grass and "dholkolmi" (*Ipomea carnea*, morning glory) and heaping piles of rotten water hyacinth around homesteads to prevent soil erosion during floods, as well as planting more trees in fields	(Reid and Alam, 2017)
Wind damage	Protecting and restoring mangroves	The cost of repairing and reconstructing houses due to combined wind and flood damage Cyclone Sidr was lowest (TK 27,043) for a site protected by mangroves compared to TK 82,246 for a site with no mangroves, and damage to trees was also lower, with 36% of trees damaged compared to 56% for the site with no mangroves	Survey Akber et al. (2018)
	Coastal shelterbelt	A 19 year old 50 m *Casuarina* shelterbelt was thought to have reduced wind speed (from 4.16 to 2.88 on a scale of 1–5) and increased the size of sand dunes (from 1.86 to 2.74)	Survey of perceived impacts Miah et al. (2013)
	Homegardens	Taller trees are grown at the boundary where they provide protection against wind but do not shade the other plants, and will not fall on the house if blown over. On the beach and hillside, owners preferred coconuts (chosen by 53% of owners), supari (45%), mango (42%) and jackfruit (25%), due to their high survival rate, strong root systems, strong stems, and low weight/light canopy which reduces the wind load on trees and prevents damage if they are blown over. However, on the mudflat island of Shahparir dwip owners preferred *Acacia* (50%), raintree (*Samanea saman*, 40%), jhau (*Casuarina equisetifolia*, 33%) and mahogany (28%) because of their strong and spreading root systems or deep taproots	Survey in Cox's Bazar Nath et al. (2015)
Water security	Conservation agriculture	Strip planting into un-tilled ground increased the water productivity of wheat by 60% compared to conventional tillage, from 1.25 to 2.06 g of grain per kg of water. Minimum soil disturbance and retention of crop residue slow evaporation, aided by the cooler temperatures under retained residue	Bell et al. (2019)
		Conservation agriculture techniques increased irrigation water productivity by 25% in rice-wheat and rice-maize systems, increasing the resilience of farmers to unpredictable rainfall patterns	Islam et al. (2019)
Food security	Conservation agriculture	A 10-year program involving over 6,000 farmers in four districts found that strip planting increased yields by up to 28% for lentil (*Lens culinaris*) and 6% for wheat (*Triticum aestivum*). Strip planting cut cultivation costs by 75%, labor requirements by 50%, irrigation water requirements by 11–33% and fuel costs by up to 85%, and increased profits by between 47% for lentil and 560% for mustard (*Brassica juncea*). Researchers on this program worked with farmers and equipment supplies to develop a lightweight reduced tillage planting machine; they estimated that if this was used by 2.5% of farmers in Bangladesh it would generate US$21–38 million per year from increased yield and reduced production costs	Bell et al. (2019)
		Increased yields for wheat and maize but not rice, lentil (*Lens culinaris*) and mung bean	Islam et al. (2019); Rashid et al. (2019)
	Fishery management and wetland restoration	The MACH project (**Supplementary Table S5**) supported local resource management organizations in re-stocking nearly 1.2 million fish (mostly juveniles) of 15 native species, which enriched fish production and biodiversity. Restoration of critical habitats can have a significant impact on catches across a much larger area, for example by excavating silted-up wetland pools in the dry season which can then be used in irrigation and to support breeding habitat for fish, and thus to contribute to food and water security. Restoration of wetland habitats and sanctuaries more than doubled fish catches, from 144 kg/ha in 1999 to an overall average of 327 kg/ha by 2007. Fish consumption of the village households around these wetlands increased by about 45% on average throughout the project period, and the landless benefited as much as larger landowners	Long term project report MACH-II, (2007a)

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TABLE 5 | (*Continued*) Key examples of evidence on the target outcomes of NbS cited in *Effectiveness of Nature-based Solutions for Addressing the Target Outcomes*

Outcome	NbS	Selected examples of outcome evidence	Reference
		Five fish sanctuaries were created in different beels (perennial water bodies) of Tanguar Haor, 12 under the CBA-ECA Project and 63 under the MACH project. These helped to increase fish supply and other aquatic resources, and enrich biodiversity, employment and tourism. Fish catches in sample areas increased from 171 kg/ha in 2013–14–277 kg/ha in 2015–16	Long term project reports: DoE, (2015) IUCN Bangladesh, (2016), MACH-II, (2007a), Winrock International (2018)
	River restoration and management	The River Halda in Chittagong is the only freshwater tidal river in the world with the right ecological conditions for the spawning of major Indian carp species, such as *Catla*, *Labeo rohita*, *Cirrhimus mrigala and Labeo calbasu*. Fish eggs and fry are gathered and sold to underpin aquaculture across the whole country. However the river is under threat from over-fishing, pollution, saline intrusion, excessive sand quarrying and unregulated construction of sluice gates for irrigation. Local people recognized the river's value and were willing to contribute their time and money to help conserve the services it provides, although there was a substantial gap between their willingness-to-conserve (Bangladesh Taka 54 million) and the value of the services provided by the river. This led the authors to recommend that community-based management, perhaps supported by a Payment for Ecosystem Services approach, should be applied to the Halda and to other rivers throughout Bangladesh	Kabir et al. (2015)

and cut the risk of erosion loss by a third (Chow, 2018). Long term community-based adaptation projects have involved local communities in planting mangroves and re-vegetating sand dunes to protect against coastal flooding and erosion (DoE, 2015; Winrock International, 2018; **Table 5**).

Despite the success of these initiatives, both existing and replanted mangroves have been extensively cleared for housing, infrastructure, agriculture and aquaculture, and degraded due to over-extraction of timber and fuelwood. Timber harvesting from mangroves was banned in 1991; this reduced the short-term benefits for local people but was expected to improve coastal protection, fish production, biodiversity and carbon storage (Abdullah-Al-Mamun et al., 2017). However such bans are not always effective, due to corruption and weak enforcement (Iftekhar and Takama, 2008; Abdullah-Al-Mamun et al., 2017; Rahman et al., 2018). Mangroves are also threatened by climate change impacts including fresh water flow reduction, sea level rise, salinity increase and storm damage (Ahammad et al., 2013; Abdullah-Al-Mamun et al., 2017; Chow, 2018; Rahman et al., 2018). Restoration should use species that are well adapted to current and future local conditions, and replanting may be necessary after a few years because there can be a high failure rate (Dasgupta et al., 2019).

Bangladesh lacks coral reefs, but one study showed that artificially created **oyster reefs** could help to reduce coastal erosion, trap suspended sediment, and support saltmarsh expansion (Chowdhury et al., 2019). However, their potential may be limited due to the high turbidity of Bangladesh's coastal waters.

Inland Flooding and Erosion

Floods, landslides, and soil erosion are severe problems in Bangladesh. In the Chittagong Hills, steep slopes, heavy rainfall, poor soil and intensive cultivation for cash crops is leading to soil erosion of over 100 t/ha/year, loss of soil nutrients, falling crop yields, landslides, and sedimentation of reservoirs (Rasul, 2009). This is exacerbated by changes in traditional slash-and-burn cultivation ('jhum'), as population pressures have shortened the fallow period during which secondary forest usually regenerates from 6 to 7 years to just 3–4 years (Gafur et al., 2003; Nath et al., 2005; Rasul, 2009). We found 16 examples of how NbS can address these hazards, and one unclear outcome.

Protecting and restoring forests plays a key role. Soil erosion was found to be three to four times lower and peak flow seven times lower in a forested catchment than a cleared catchment (Gafur et al., 2003), and modelling indicates that trees with deep tap roots could prevent landslides on slopes less than 70° (Islam and Rahman, 2019). **Agroforestry** can reverse soil loss, as the soil-formation rate can exceed the erosion rate (Rasul, 2009). Contour planting (planting trees across slopes) can be particularly effective, and can increase yields, income and food security (e.g., in the MACH project, MACH-II, 2007a,b). In the Haor Basin, community-based adaptation initiatives are restoring **freshwater swamp forests** and planting **trees on embankments** to protect villages on 'kanda' (raised land that becomes islands when the seasonal wetlands flood) from erosion due to wind-driven wave action (DoE, 2015; IUCN Bangladesh, 2016;

Winrock International, 2018). Villagers also grow low plants and heap piles of rotten water hyacinth around homesteads to prevent soil erosion during floods (Reid and Alam, 2017).

Wind Damage

Cyclones and other tropical storms cause damage from high winds as well as flooding. Trees can help to protect against this, although they can also cause damage if they are blown over (Nath et al., 2015) and, conversely, forests and mangroves can themselves be damaged in storms (Dasgupta et al., 2019). We found six positive outcomes of NbS for addressing wind damage.

Protection by **mangroves** in the Sundarbans reserve significantly reduced the cost of repairing houses due to combined wind and flood damage (Akber et al., 2018), and **coastal shelterbelts** of *Casuarina* trees were also perceived to be effective in reducing wind speed and increasing the size of sand dunes (Miah et al., 2013). In **homegardens**, a survey found that taller trees are grown at the boundary where they provide protection against wind but do not shade the other plants, and will not fall on the house if blown over (Nath et al., 2015). On exposed beach and hillside locations, species such as coconuts were preferred due to their strong stems, low weight, and light canopy, which reduced the wind load on trees and reduced damage if they fall, while species such as *Acacia* were chosen on mudflats because of their strong and spreading root systems.

Heatwaves

Only two of the studies retrieved in our review referred to NbS for protecting against heat, and neither provided robust evidence. In the Haor basin, villagers value the forest for providing shade and cooling the air, and they are also planting more fruit trees at their homesteads for shade during heatwaves (Reid and Alam, 2017). However, native trees such as mango and jackfruit are susceptible to hailstorm damage, so they are being replaced with fast-growing and storm-resistant non-native timber trees such as teak, eucalyptus and acacia (Reid and Alam, 2017), so this action might not be classified as a NbS as there could be adverse biodiversity impacts. On the coast, local people planted salt-tolerant mangrove trees on land degraded by a saline storm surge where all the other trees had died, to provide protection from summer heat (Imam et al., 2016). The evidence in these studies was based on the perception of the villagers who carried out the intervention.

Water Security

Although Bangladesh experiences extremely heavy rainfall in the monsoon season, droughts and water shortages are a growing problem in the dry season (Sayed et al., 2020), due to climate change, over-abstraction of water for human use, soil degradation, water pollution and salinization.

Ten interventions were associated with outcomes for water security: 8 positive and 2 unclear. Four of the positive outcomes were from **conservation agriculture**, where improving soil structure and retaining crop residue can reduce evaporation and enhance soil water infiltration and storage, reducing the need for irrigation (Alam et al., 2017; Bell et al., 2019; Islam et al., 2019; Sayed et al., 2020). For example, these techniques increased the water productivity of wheat by 60% (Bell et al., 2019) and increased irrigation water productivity by 25% (Islam et al., 2019), increasing the resilience of farmers to unpredictable rainfall patterns.

Rainwater harvesting mainly involves engineered options such as installation of rooftop tanks. However, a study in the Barind Tract described how re-excavation of silted-up ponds and beels (permanent wetland waterbodies) allowed rainwater storage and infiltration to recharge a depleted aquifer, so that the community now has water all year round rather than only for 4–5 months of the year (Rahaman et al., 2019).

Four interventions addressed water pollution, which seriously affects both surface water and groundwater in Bangladesh. We found evidence that **constructed wetlands** could be used to help tackle the problem of groundwater pollution by arsenic, which affects over 1.5 million wells used by 35 million people in Bangladesh. Experiments indicate that water from wells could be passed through a series of wetlands containing river sand and planted with bulrush (*Typha latifolia*), which would reduce arsenic concentrations to the WHO safe limit (Schwindaman et al., 2014). Two studies mentioned the potential role of water hyacinth for removing water pollution in inland wetlands, but neither provided robust data (MACH-II, 2007a; MACH-II, 2007b; Reid and Alam, 2017).

Food Security

Food security accounted for 43 of the 140 positive outcomes identified in the review, and nine mixed, negative, neutral or unclear outcomes. Conservation agriculture was the most frequently cited NbS, but others included agroforestry, homegardens, wild food in protected forests, and the role of protected and restored mangroves and inland wetlands in supporting fisheries.

Smallholders face severe challenges in Bangladesh, as soil fertility is declining, rainfall is becoming more unpredictable and the cost of agrochemicals, irrigation and fuel is increasing. **Conservation agriculture** has been extensively studied as a way of improving soil health and water storage capacity, with a focus on adapting the approach for the rice-based rotations that predominate in Bangladesh. The key benefit for food security is that conservation agriculture enables inputs of mineral fertilizers and irrigation to be reduced without loss of yield (Islam et al., 2011; Alam et al., 2020), and this in turn increases profitability (Aravindakshan et al., 2015; Gathala et al., 2016). The papers stated that the key is to apply integrated science-based approaches that combine conservation agriculture techniques with reduced (but not zero) use of synthetic fertilizers, to maintain yields whilst cutting costs (Islam et al., 2011; Begum et al., 2018).

For example, a 10-year program involving over 6,000 farmers found that strip planting increased yields by up to 28% and cut cultivation costs by 75%, labor requirements by 50%, irrigation water requirements by 11–33% and fuel costs by up to 85%, increasing profits by 47–560% for different crops (Bell et al., 2019) (**Table 5**). Researchers on this program worked with farmers and equipment supplies to develop a lightweight reduced tillage planting machine; they estimated that if this was used by 2.5% of farmers in Bangladesh it would generate US$21–38 million per year from increased yield and reduced production costs.

Similarly, **integrated pest management**, which combines non-chemical methods of pest control (such as manual removal of pest eggs, and provision of perches for birds) with reduced application of pesticides at the economic threshold level (at which the value of the crop destroyed exceeds the cost of controlling the pest), can increase yields and save money compared to conventional heavy pesticide use, while reducing pollution and health impacts from pesticide poisoning (Alam et al., 2016).

Agroforestry offers a method of increasing food, fuel and timber production despite shrinking available land and a growing population, by making use of vertical space (Hanif et al., 2015). It can help to reduce the negative environmental impacts of food production such as soil erosion, water pollution and (if native tree species are used) biodiversity loss, and can help to restore degraded land. The first two Community Forests in Bangladesh were established on deforested land in the Chittagong Hill Tracts that the Forest Department had already tried to replant several times but without success, due to lack of maintenance and encroachment for farming by local people. The land was given to local landless or almost landless people, who first planted eucalypts and fruit trees, and later shifted to native forest species. Sale of tree products improved their incomes and enabled them to diversify to other income sources, relieving pressure on local natural forests, but as they become less dependent on forest resources it remains to be seen whether this reduces their incentive to conserve and replant the community forest (Mohammed et al., 2016). Agroforestry could also potentially help to restore land that has been degraded by saline intrusion, although there is a risk that it could exacerbate the problem by concentrating salts in the root zone (Wicke et al., 2013).

Homegardens are widespread and play a vital role in food security in Bangladesh. One study found that applying a year-round rotating system of different vegetables and fruit trees, chosen through a participatory approach using local traditional and indigenous knowledge, could more than double annual production, improve nutrition, increase household income, alleviate poverty and provide employment and empowerment for female family members (Ferdous et al., 2016). In dense urban areas, rooftop gardens can play a key role; 35% of properties surveyed in one part of Dhaka had rooftop gardens providing a wide range of fruit and vegetables (Zinia and McShane, 2018).

Floating gardens are a traditional approach to cultivation in some wetland areas, and have more recently been introduced to the Haor basin. Mats of floating wetland vegetation (mainly water hyacinth, which is an invasive non-native species) are used as the base for vegetable gardens in the wet season when farmland is inundated, and the rotting mat is then used to fertilize the soil in the dry season when the water recedes. This can help families to produce food all year round, reducing the need for unsustainable harvesting of wild resources to sustain livelihoods (Irfanullah et al., 2008).

Wild food resources are important to many low-income households. People living closer to the Sundarbans mangrove forest have higher levels of dietary diversity, partly due to direct consumption of bushmeat, although the authors noted that bushmeat harvesting for subsistence often coexists with commercial bushmeat trades that may threaten endangered species (Baudron et al., 2019).

Forest ecosystem services can also contribute positively to food production in nearby cropland. A survey of 275 households in the Chittagong Hill Tracts found that households closer to forests were more likely to have homegardens and own livestock, and had higher fruit consumption and higher dietary diversity as a result. The nearby forest was thought to support crop production in homegardens through maintenance of soil fertility, micro-climate regulation, and pollination (Baudron et al., 2019). The benefits of these existing forests are a proxy for the benefits that would continue to be delivered through protecting and/or restoring the forests in future. However, protecting or restoring ecosystems such as forests and mangroves can reduce the area available for food production, potentially leaving a 'food gap' where an area is unable to produce enough food to meet local demand (Hoque et al., 2020). To address this, it has been suggested that, for example, coastal areas targeted for NbS to provide hazard protection could receive food subsidies from surplus food produced in other areas further inland (Hoque et al., 2020).

Sustainable fishery resource management has long been practiced by communities in the freshwater wetlands of Bangladesh. Community-based projects have brought all the remote beels (permanent pools within the seasonal wetland) under one management practice, allowing the community to harvest fish to a sustainable level under certain conditions determined by biodiversity conservation goals (IUCN Bangladesh, 2016). This supports poor people's rights and access to resources, as well as boosting food and nutrition security, employment and biodiversity. In the Tanguar Haor project, endorsement of a 'core zone and buffer zone' approach by the government was a significant achievement which established the rights of the poor to access fisheries across the wetland, and reduced illegal harvesting (IUCN Bangladesh, 2016). In addition, the projects build the capacity of the community by developing income generating and community management skills, raising awareness of wetland resource issues and understanding the value of biodiversity conservation (IUCN Bangladesh, 2016). Community-managed fish sanctuaries have also been created in deeper parts of the wetland, where fishing and collection of other aquatic resources are restricted or totally prohibited. In Hakaluki Haor, fish sanctuaries were established in two beels by using katha (bamboo and tree branches) to create breeding grounds and food supply for the fish and protect from illegal fishermen. This helped to increase fish catches from 171 kg/ha in 2013–14 to 277 kg/ha in 2015–16, while enriching biodiversity, employment and tourism (Winrock International, 2018; **Table 5**). The MACH project also supported local resource management organizations in re-stocking nearly 1.2 million fish of 15 native species, and restoring critical wetland habitats. This enriched fish production and biodiversity, more than doubling fish catches, and increasing fish consumption in local villages by about 45%, with landless people benefiting as much as larger landowners (MACH-II, 2007a).

River protection, restoration and sustainable management can sustain and enhance essential services including provision of fish and fresh water. For example, the River Halda is the only river in the world where major Indian carp species can spawn (Kabir et al., 2015). Fish eggs and fry are gathered and sold to underpin aquaculture across Bangladesh. However, the river is under threat from over-fishing, pollution and other human activities. A survey found that local people were willing to contribute their time and money to help conserve the river, although their willingness-to-conserve was less than the value of the services it provided (Kabir et al., 2015). The authors recommended that community-based management of rivers could be supported by a Payment for Ecosystem Services approach.

Wood, Fuel, and Non-Timber Forest Products

Forests are a vital source of wood, timber and other products for local people in Bangladesh, although over-exploitation has led to widespread forest degradation (Yong Shin et al., 2007). There were 19 outcomes related to this, of which three were negative, all from a study of the impacts of forest protection in the Chittagong Hills (Miah et al., 2014). This found that local households generated an average revenue of Bangladesh Taka 13,473 per year from gathering timber, firewood, bamboo, medicinal plants, bushmeat and nuts, as well as being dependent on these goods for their own use. Dependence on the forest for medicinal, religious and food purposes was felt to be non-negotiable, while some people were prepared to forego extraction of timber, bamboo and vegetables if appropriate cash compensation or alternative livelihoods were provided. The authors recommended that providing secure land tenure for the indigenous communities could help to establish more sustainable management of the forest resources, which were perceived to be severely degraded due to over-extraction.

Many NbS interventions attempt to manage this conflict by implementing participatory sustainable management approaches. One example is the **social forestry** program, which was launched by the Bangladesh Forest Department in 1989 to support forest restoration, agroforestry, village woodlots, and roadside plantations (Rahman et al., 2015). Most of the roads in south-western Bangladesh are lined by social forestry plantations that are protected and managed by local landless and land-poor people, who are allowed to gather fuelwood and receive 40% of the proceeds after felling (Rahman et al., 2015). Similarly, in Hakaluki Haor, the 'green belts' of indigenous tree species established on embankments provide fuel wood and local economic benefits as well as reducing flood risk (DoE, 2015). Local communities are also allowed to gather branches for fuelwood from restored swamp forest when it is sufficiently mature, but there are agreements not to fell the trees (MACH-II, 2007a; IUCN Bangladesh, 2016).

Conflicts between different beneficiaries are illustrated by a case study of Nijhum Dwip, an uninhabited mudflat island that was planted with mangroves by the Forest Department and was later declared a National Park, due to its value for migratory birds (Iftekhar and Takama, 2008). However, around 700 households who had been displaced from their homes due to river erosion gradually settled on the island, leading to illegal encroachment and extraction of timber from the forest. Around a quarter of people interviewed on the island were highly dependent on forest resources, especially women and low-income households, and the forest also attracted tourists who boosted the local economy. Yet many residents wanted to convert the forest to agriculture, aquaculture or commercial forestry. Despite this, and in apparent contradiction, almost all thought that the forest should be better protected and that new mangroves should be planted. It was suggested that adaptive co-management involving local people could help to resolve these conflicts and trade-offs through sustainable use and equitable distribution of benefits.

Cultural Outcomes

We found only six cultural outcomes in the review, three related to the value of protecting the Sundarbans mangroves for aesthetic value, recreation and cultural heritage and spiritual inspiration (Rahman et al., 2018). Cultural heritage was also an important benefit of protecting the River Halda, which inspired local festivals, characteristic Sampan boats and the Halda Fada songs (Kabir et al., 2015). However, trade-offs were found for a shelterbelt of casuarina trees, which increased the aesthetic value of the area and its attractiveness for tourism, but also acted as a location for increased anti-social behavior such as theft (Miah et al., 2013). Finally, the CREL (Climate Resilient Ecosystems and Livelihoods) project (**Supplementary Table S5**) was estimated to deliver US$53 million of tourism and cultural services (Winrock International, 2018).

Socio-Economic Outcomes

There were 52 outcomes reported for socio-economic benefits: 50 positive, one negative and one mixed. These included local economic benefits (15), employment (8), tourism (8), rights, empowerment and inequality (8 positive, 1 mixed), education and training (6), and social cohesion, governance and engagement (5 positive, 1 negative).

Local economic benefits include increased profits from implementation of conservation agriculture or agroforestry, income from use of sustainably harvested natural resources (including social forestry), and support for livelihoods provided through community-based management programs. Tourism also provides benefits for local economies, through opportunities for eco-tourism or enhanced attractiveness of tourist destinations. For example, a coastal shelterbelt was perceived to have increased tourist visits to a beach, due to provision of shade and increased attractiveness (Miah et al., 2013). Employment benefits came from jobs created through habitat restoration and protection, or livelihoods sustained through better management of resources. Out of the 154 interventions, 42 explicitly targeted poverty reduction although only 12 of these provided clear evidence that poverty had been reduced as a result of the intervention, while one reported no effect and five had unclear evidence. There was one mixed outcome for a social forestry initiative where poor people were excluded because the land was illegally occupied by local elites (Muhammed et al., 2008).

Out of the 50 positive socio-economic outcomes, 33 came from the community-based initiatives described in the grey literature reports (**Supplementary Table S5**). These were generally initiated by government departments with support from international development or conservation NGOs. They aimed to conserve biodiversity and provide sustainable livelihoods for local people, through empowering and engaging the community to manage their wetland and forest resources sustainably, with a strong focus on reducing poverty and inequality (DoE, 2015). Community organizations were established, and local people were offered training in natural resource management and conservation techniques and supported to develop diversified livelihood options, including through skills training and establishment of micro-credit loan schemes. For example, the CREL project provided training and support for enterprise development to

60,000 households, of which 51,400 reported that they had adopted more resilient agricultural practices, and 38,500 were estimated to have enhanced their incomes by a total of over US$5 million. Over 8,000 poor women (73% of livelihood beneficiaries) were empowered through financial training, helping them to improve access to services and credit, increase asset ownership and play a greater role in decision-making. CREL also helped 45 community management organizations to develop governance and financial capacity, with 35 of these (79%) reporting that they were able to become recognized local implementing partners for government and donor programs, and 23 (52%) becoming able to generate sustainable income from charging visitors to enter their protected areas (Winrock International, 2018).

These capacity building activities were accompanied by community-led restoration and protection of the ecosystems on which the communities depend, such as restoring swamp forests, excavating dried up wetlands and re-stocking with fish, with community guards protecting the restored forests from illegal exploitation. Co-management approaches were established, along with mechanisms for sharing the benefits from natural resources, such as by distributing 60% equally to households involved in the initiative, 25% to the community-based organization, and 15% to the government as revenue (IUCN Bangladesh, 2016).

Community management can help to build social cohesion and provide opportunities for engagement, education and cultural enrichment. For example, community organizations established two bird sanctuaries in Tanguar Haor, and employed community guards to protect them from illegal hunting, providing opportunities for eco-tourism development. A nature club was also established to engage young people in conservation and raise awareness on the importance of maintaining the ecological integrity of the wetland (IUCN Bangladesh, 2016). In the CBA-ECA project, local people started to feel a sense of pride in having a bird sanctuary in their neighborhood, and this empowered them to resist illegal hunters (DoE, 2015).

Several studies noted the importance of protecting intact ecosystems in order to sustain the flow of benefits on which many households depend, especially the most vulnerable. For example, the Sundarbans mangroves were estimated to provide public goods worth US$ 1,135 per ha each year, greater than the net economic return from shrimp farming at US$ 713 per ha (Rahman et al., 2018). A land use model of the Lower Meghna River Estuary estimated that loss of forests and mangroves due to urban expansion resulted in loss of ecosystem services worth US$118 million from 1988 to 2018, and continuing with business as usual or prioritizing economic development will lead to further losses of US$41 or US$16 million respectively, while protection and restoration will deliver an additional US$131 million of benefits (Hoque et al., 2020).

Biodiversity Benefits

NbS should, by definition, support and preferably enhance ecosystems and their biodiversity. However, ecological outcomes were only explicitly reported for 28 interventions (all positive), and only four of these had robust evidence. Often the benefits were reported in terms of species richness or presence of iconic species, and in anecdotal terms with no clear methodology or baseline.

We aimed to screen out any interventions for which there was no obvious pathway for delivering ecological benefits, which would not be defined as NbS, but in several cases this was not clear. For example, social forestry plantations were found to contain 36 tree species from 17 families, but 94% of the biomass was from just four fast-growing timber species (Rahman et al., 2015). Similarly, it was reported that homegardens have the same tree species diversity as natural forests, but the species composition is different to natural forests, with a bias towards fruit, nut and ornamental trees, some of which are not native species (Bardhan et al., 2012). The ecological outcomes of these interventions would depend on the most likely alternative use of the land.

Nevertheless, we did find evidence of the role of NbS in supporting the biodiversity of the unique and threatened forests, mangroves and wetlands in Bangladesh. Many of these were related to the community-based management projects, which protect threatened habitats and species while also providing jobs and eco-tourism opportunities (**Supplementary Table S5**). For example, protection and restoration of Baikka Beel within Hail Haor resulted in an increase in wintering water bird populations from about 300 birds of 16 species in 2004 to 7,200 birds of 35 species in 2007 (MACH-II, 2007b). Similarly, rapid assessments in Nuniarchhara mangrove forest found 24 wildlife species in 2011–2013 (18 birds, 2 mammals, 1 reptile, 3 amphibians) compared to 14 bird species in 2007–2010, although it is not clear whether the earlier assessment looked for non-bird species. The report also notes that seven species of kingfishers were observed in more recent years, which indicates abundance of native fish, as well as some rare species such as the fishing cat and monkey, and Purple Swamphen (Kalim) bird.

Enabling Factors for Successful Implementation of Nature-based Solutions

Several enabling factors were reported to influence the successful implementation and governance of NbS. We have classified these into five groups: participatory delivery incorporating local knowledge; strong, transparent and equitable governance; access to finance; secure land tenure; and practical support such as training.

Participatory Delivery

The literature identifies a long tradition of research and implementation of participatory and pro-poor approaches to natural resource governance in Bangladesh, including harnessing local and traditional knowledge (Alam et al., 2016; Ferdous et al., 2016; Dasgupta et al., 2019), targeting interventions towards landless or land-poor households (Muhammed et al., 2008; Miah et al., 2014; Rahman et al., 2015; Ferdous et al., 2016), and working with local communities to protect and manage resources (Rahman et al., 2015). It was suggested that participatory co-management could help to resolve trade-offs and conflicts between beneficiaries or between different outcomes (Iftekhar and Takama, 2008).

The grey literature describes long-term projects (**Supplementary Table S5**) that aim to build capacity for community-based management of natural resources, by establishing community organizations and supporting vulnerable people to adopt diversified and sustainable livelihoods (*Socio-Economic Outcomes*). They show how participatory approaches are crucial for engaging and motivating the community to protect and sustainably manage natural resources (DoE, 2015). Adopting a co-management approach for managing ecologically important wetlands and forests during these initiatives was said to trigger a cultural and policy shift for the government of Bangladesh, from a top-down protection approach towards working with local communities to address the underlying drivers of over-exploitation of natural resources (Winrock International, 2018). Although the benefits of these approaches are self-reported by the project implementers, they hold considerable promise for supporting livelihoods and empowering the vulnerable while protecting critically endangered habitats and species. However, the study methods and outcomes were not always clearly reported.

We found one example of an approach that was implemented without sufficient participation. Attempts to train landless indigenous people to establish contour hedgerows to help them cultivate degraded forest land in the Chittagong Hill Tracts failed, because they found the system too complex and labor-intensive, it used unfamiliar hedgerow species and did not leave enough available land to meet their needs for subsistence cereal crops (Nath et al., 2005). International guidance has now been developed to help define best practice for participatory NbS (IUCN, 2020), including standard reporting criteria, which should help to improve future outcomes.

Governance

A common theme in the literature is that poor governance and corruption frequently undermines the effectiveness of interventions (Iftekhar and Takama, 2008; Abdullah-Al-Mamun et al., 2017). For example, a participatory forestry initiative in Tangail Forest Division showed a bias towards allocating land to local elites who had illegally encroached into forest areas, rather than landless people as intended, due to corruption of forest officials (Muhammed et al., 2008). Rahman et al. (2015) recommend stricter implementation of operational rules, strengthening of institutions for regular monitoring, and increased authority to implement sanctions against violators, to enhance the outcomes of roadside social forestry.

The cost of corruption has been estimated in a study of the Chittagong Hill Tracts, where smallholders must bribe local forestry officials in order to get a license to sell timber from agroforestry, as well as paying bribes at all the checkpoints established to (in theory) control illegal felling. Without paying the bribes of Taka 150 per cubic foot of timber, roughly 20% of the market price, the profitability of agroforestry would double. Setting up local collectives would enable smallholders to get a fairer price for agroforestry products by weakening the market dominance of large traders and middlemen (Rasul and Thapa, 2007). Strong, transparent and fair institutions that focus on empowering the vulnerable can ensure that the benefits of NbS flow to those most in need.

Finance, Land Tenure, Training, and Other Support

Although NbS can offer more cost-effective solutions than alternatives in the long term, when all public and private costs and benefits have been taken into account, governments and other funding agencies may need to provide practical and financial support to enable the transition to NbS in a way that meets local needs. For example, farmer training and knowledge of pests and beneficial insects is crucial to application of integrated pest management (Alam et al., 2016), and farmers on the plains of north-west Bangladesh were more likely to adopt conservation tillage if they had access to an agricultural extension office to provide unbiased advice and training (Aravindakshan et al., 2015).

Similarly, in the Chittagong Hills, subsistence farmers who could benefit from agroforestry to stabilize and regenerate the eroding soil (**Section 3.3.3**) face short term barriers including a high cost of borrowing, small size of land holdings, and pressure to produce sufficient crops to feed their families. Agroforestry does not produce economic returns for the first 5 years for fruit, or 10–12 years for timber, while shifting 'jhum' agriculture produces crops in just a few months (Rasul and Thapa, 2006; Rasul, 2009). Also, most farmers do not have secure land tenure, as the forests were nationalized during colonial times, and this discourages long term investments and prevents access to credit for covering initial costs. As a result, farmers continue to practice slash-and-burn cultivation on common land, avoiding the costs of nutrient depletion and soil erosion in the short term by shifting to new locations, but undermining their livelihoods in the long term. This implies that governments and other funding agencies need to enable a shift to more sustainable farming practices by providing financial incentives (such as Payment for Ecosystem Services), access to credit, secure land tenure or inheritable land use rights, and practical training and support (Rasul and Thapa, 2006; Rasul, 2009).

DISCUSSION

Evidence on the Effectiveness of Nature-based Solutions for Addressing Societal Challenges in Bangladesh

A wide range of NbS are being implemented in Bangladesh, including protection and restoration of forests, mangroves, and wetlands; conservation agriculture; agro-forestry; and participatory forest, fishery, and wetland management. We found robust evidence on the benefits of these activities for reducing vulnerability to cyclones, storm surges, floods, landslides, and salinization, and helping communities adapt to sea level rise, water shortages, high temperatures and extreme rainfall. Carefully designed and managed NbS can sustain livelihoods and reduce poverty and social inequality, by ensuring that the benefits flow to poor, landless and disadvantaged members of the community. In summary, we found examples of how NbS can address climate change and natural hazards while contributing to almost all the Sustainable Development Goals (**Supplementary Table S7**).

NbS must support or preferably enhance biodiversity (Seddon, Smith et al., 2021), and this could help address degradation of natural habitats in Bangladesh (Bardhan et al., 2012), enhancing the delivery of ecosystem services. For example, we found evidence that species diversity is linked to greater resilience of mangroves to pests and diseases (Dasgupta et al., 2019), greater soil carbon sequestration (Islam et al., 2011) and improved opportunities for eco-tourism (IUCN Bangladesh, 2016). However, most studies did not report evidence of biodiversity benefits, and those that did were largely confined to reports of species richness for a limited number of taxa (mainly birds). In some cases, it was not clear whether biodiversity benefits had been achieved – such as for social forestry plantations that used mainly fast-growing non-native timber species (Rahman et al., 2015). To determine whether biodiversity benefits arise, it is important to report on the baseline or counterfactual scenario, i.e., the previous use of the land, and the likely future use in the absence of the NbS. Ideally, there would be a survey of biodiversity before and after NbS implementation, covering the abundance and richness of multiple taxa such as birds, mammals, reptiles, invertebrates, amphibians, higher and lower plants, and fungi, or, if this is too costly, at least a basic survey to identify the presence or absence of species of conservation concern.

There are gaps in the evidence base in Bangladesh for certain types of NbS, including urban green infrastructure. Evidence from other low to middle income countries could be useful to assess the relevance of these solutions in Bangladesh. In addition, data from other countries with similar ecosystems can help to refine and validate the results of the search for Bangladesh. For example, studies in Florida and New Zealand suggest that the primary benefit of mangroves for coastal flood protection may be in reducing the velocity rather than the height of storm surges (Krauss et al., 2009; Montgomery et al., 2018). However, Bangladesh also provides useful evidence which could be relevant to other countries, such as on mangrove and wetland restoration, community-based resource management, homegardens, floating gardens and application of conservation agriculture techniques to rice-based cropping systems.

Despite the strong potential for NbS to deliver multiple benefits, there are also some limitations. For example, NbS alone may not deliver complete protection from coastal and river flooding – in some areas it will be necessary to combine NbS with engineered defenses and effective hazard warning systems. However, because NbS can reduce hazards such as wave height and velocity, the engineered elements of such hybrid approaches may be smaller and cheaper (e.g., lower embankments), and NbS can also strengthen, shelter and protect infrastructure such as levees so that it is cheaper to maintain and less likely to fail (King and Lester, 1995; Thornton et al., 2019).

Although 62% of outcomes reported in the academic evidence were based on strong evidence, many projects were only reported via grey literature such as project reports, which contained only weak evidence on outcomes. The evidence base could be strengthened by using consistent methodologies to monitor and report on the outcomes of NbS over time; gathering robust quantitative or qualitative data that shows the impacts relative to a baseline or counterfactual and takes account of confounding factors; recording synergies and trade-offs between outcomes; clearly describing governance arrangements, the mode of community participation and social distribution of benefits; and recording measurable outcomes for biodiversity.

Strengths and Limitations of the Review

As far as we are aware this is the first systematic review on the effectiveness of NbS for addressing societal challenges in Bangladesh. It shows how the methodology developed for the global review by Chausson, Turner et al. (2020) can be adapted to focus in more detail on a single country and more comprehensively cover additional outcomes including climate change mitigation and development goals.

There are several limitations: we did not cover articles in languages other than English; and we excluded 236 papers at the abstract screening stage that did not refer to evidence on the effectiveness of NbS in the abstract, which could contain relevant evidence in the main text. However, our co-authors in Bangladesh can confirm that almost all projects pertinent to NbS prepare their most important reports in English, since these are funded by various development partners. Other project outputs such as reports on specific activities, guidelines for community-based organizations, case studies and communication materials are usually prepared in the local language (Bangla), but these are not suitable sources of information for our review. In addition, almost all peer-reviewed journal articles are in English.

We covered only studies of NbS interventions in Bangladesh. Studies on similar NbS in other countries could also be useful. For example, Chausson et al., 2020 found 14 papers on mangroves for coastal protection in other countries and 13 papers on NbS in South and South-east Asia, many of which may be relevant to Bangladesh.

Scaling up High Quality Nature-based Solutions in Bangladesh

Our review showed the importance of protecting irreplaceable natural assets such as forests and wetlands in planning policies, and recognizing the non-market benefits they deliver. However, the integration of NbS into policy is currently patchy and inconsistent. For example, a review of key national and sectoral development and climate change policies found that although Ecosystem-based Adaptation (EbA) is considered in most of them, especially at the top strategic level, it is largely ignored at the policy formulation and implementation stage where priority is given to engineered approaches such as concrete dams and embankments (Huq et al., 2017). Only 38 out of 329 climate change adaptation projects reviewed were related to ecosystem interventions and of these 14 were river dredging, and the rest were mainly concerned with commercial forestry, neither of which are NbS as they may have adverse biodiversity impacts. All sectoral development policies except the coastal sector largely ignored the potential for EbA, with climate change adaptation and ecosystem approaches being seen as competing rather than complementing one another. The review concluded that there was an institutional and cultural bias towards hard engineering adaptation options, and lack of awareness of the potential of NbS/EbA amongst policymakers, compounded by top-down decision-making, bureaucracy, lack of stakeholder engagement and corruption.

However, more recent analyses showed that Bangladesh's policy documents, strategies and plans do involve certain elements and approaches of NbS (Tasnim et al., 2020; Irfanullah 2021a) and that Bangladesh is showing increasing policy interest in NbS (Irfanullah, 2020). Several important, practical suggestions arose from a consultation on NbS for development planning in Bangladesh co-organized by Bangladesh Planning Commission, involving government agencies, researchers and practitioners (ICCCAD, 2020). These included: 1) NbS should be incorporated in Bangladesh's 5-year development plans; 2) use of NbS for mitigation and adaptation to climate change should be included in Bangladesh's Nationally Determined Contributions (NDC) in detail; 3) government's project design guidelines should include NbS so that ecosystem-based approaches are always considered in development projects; 4) a NbS database should be created to encourage a deeper understanding of NbS and aid identification of good practice; 5) opportunities to incorporate NbS in the agriculture sector should be explored, to reduce damage to biodiversity from the food supply chain; and 6) local people should be at the core of NbS planning and implementation.

Similarly, a systematic analysis of twenty policy documents in the development, climate and environment sectors found that although only one used NbS terminology, there was a growing emphasis on the protection and management of natural ecosystems using concepts such as ecosystem-based adaptation, ecosystem services, and green building (Islam et al., 2021). Nevertheless, there was a lack of implementation guidelines, financial support and mechanisms for monitoring and evaluating NbS initiatives, and a need for greater inter-ministry cooperation; national funding support; a national promotional campaign; more evidence-based research and capacity-building; and greater involvement of youth, marginalized people, and women.

Implications and the Way Forward

We have identified many promising NbS initiatives in Bangladesh, but there is potential to achieve much greater benefits by scaling these actions up across the country, and adopting best practice to maximize the benefits and minimize trade-offs. Based on the findings of this review, we identify four priority areas for action on NbS by Bangladesh, which are also likely to be applicable to other low and lower-middle-income countries.

Strengthening the Evidence Base and Integrating It Into Policy

There is an opportunity to capitalize on recent interest in NbS, both globally and in Bangladesh, and to promote evidence-informed policy and practice to influence nature conservation and climate resilience. The evidence that we have compiled on the effectiveness of NbS interventions, such as the creation of coastal green belts with mangroves over the last 56 years, protection of World Heritage and Ramsar Site the Sundarbans, and sustainable management of wetlands over the last 22 years, is a good starting point, but we also recommend strengthening this evidence base through a more systematic approach to monitoring, evaluating, and reporting the process and outcomes of future NbS projects. Nevertheless, our analysis can help Bangladesh to effectively incorporate nature conservation and ecosystem-based approaches in implementing its current plans in the short-term (e.g., *8th Five-Year Plan 2020–2025*), medium-term (e.g., *Perspective Plan of Bangladesh 2021–2041*), and long-term (e.g., *Bangladesh Delta Plan 2100*). The evidence on NbS effectiveness that we have compiled can also help the country to take pragmatic steps in implementing the NDC and National Adaptation Plan (NAP) (under preparation) as well as any plans developed in response to the *Post-2020 Global Biodiversity Framework* to guide biodiversity conservation through 2050.

Nature-based Solutions for Economic Recovery

As the world is trying to focus on post-pandemic recovery, the International Labour Organization has advocated the potential for NbS to boost economic recovery by increasing green employment opportunities (WWF and ILO, 2020). Under its Nature-based Recovery Initiative, IUCN has been working with its Members and partners to create evidence to support governments to invest at least 10% of overall investments in nature and to ensure that economic investment in the post-COVID era doesn't cause further harm to nature and livelihoods (IUCN, 2021). Bangladesh is in a good position to harness this opportunity, given its long experience of implementing NbS interventions as community-based management of natural resources, community-based adaptation, and co-management of protected areas. The economic recovery potentials of different NbS interventions in a wide range of ecosystems of the country could be investigated and incorporated in the national COVID recovery plans.

Urban Nature-based Solutions

Our world is urbanizing exponentially, and 68% or 7 billion people could live in cities and towns by 2050 (WEF, 2020), yet experience of urban NbS in Bangladesh is relatively limited. Some recent initiatives bring together policies and practices on ecosystem-based approaches in urban areas, such as the Global Commission on Adaptation (www.gca.org), the Network Nature (www.networknature.eu) of the European Union, and the BiodiverCities by 2030 initiative of the World Economic Forum and the Government of Colombia (www.weforum.org). In Bangladesh, with highly vulnerable coastal towns and increasing climate-induced displacements, urban local government institutions and development partners should make NbS an integral part of urban development strategies and plans (Irfanullah, 2021b). NbS interventions in urban settings should restore and manage urban ecosystems and biodiversity, help to address conflicts over natural resources, and ensure social equity within the expanding urban slums. Local institutions and communities should either lead or be appropriately and sufficiently involved in planning, executing, and monitoring NbS in towns and cities.

Nature-based Solutions Guidelines

It is important to understand the scope, effectiveness and limitations of NbS to avoid any miscommunication, misuse and misinterpretation (Irfanullah, 2021a). Funding and implementing agencies and other stakeholders should abide by the available standards and guidelines when designing, implementing, and scaling up NbS initiatives. The *IUCN Global Standard for NbS* (IUCN, 2020) brings together experience from 100 countries. It guides stakeholders to co-define societal challenges so that they can co-design suitable NbS at an appropriate scale, and checks that NbS are economically feasible, provide sufficient biodiversity and human well-being benefits, involve all stakeholders equitably, and manage trade-offs. Similarly, Seddon et al. (2021) urge stakeholders to follow four guiding principles: NbS are not a replacement for the rapid decarbonization of the economy; they should involve a range of terrestrial, freshwater and marine ecosystems; they are designed, implemented, managed and monitored by or in partnership with Indigenous peoples and local communities; and they should provide measurable benefits for biodiversity.

CONCLUSION

We have found that a range of NbS are already being implemented in Bangladesh, and these are helping to address interlinked societal challenges including disaster risk reduction, climate change adaptation and mitigation, biodiversity loss and other sustainable development goals. There is robust evidence that protecting and restoring forests and mangroves helps to protect communities and property from cyclones, floods and landslides. Together with conservation agriculture, agro-forestry, and participatory fishery and wetland management, these NbS help communities to reduce their vulnerability to the impacts of climate change such as sea level rise, water shortages, high temperatures and extreme rainfall. Understanding the benefits of NbS can help to make the case for protecting Bangladesh's remaining high value natural assets, including the Sundarbans mangroves and Chittagong hill forests, as well as implementing more sustainable agricultural practices such as agro-ecology and agroforestry in the farmed landscape. Carefully designed and well-governed NbS can also help to deliver development benefits by sustaining livelihoods, boosting local economies, strengthening institutions, and reducing poverty and social inequality. In summary, NbS support an integrated approach to delivering multiple Sustainable Development Goals and provide the foundation for a Green Recovery from the COVID-19 pandemic.

However, NbS need to be implemented carefully and in line with good practice guidelines in order to manage trade-offs and secure multiple long-term dividends for both nature and people. Key enabling factors are a participatory approach that incorporates local and traditional knowledge; strong and transparent governance and community institutions; and a focus on empowering the vulnerable and equitably distributing the benefits to those most in need. Attention is also needed to ensure that NbS deliver genuine benefits for biodiversity, thus helping to sustain resilient ecosystems which can underpin health and prosperity in the long term.

The review revealed an evidence gap on urban green infrastructure, a lack of strong evidence on biodiversity outcomes, and inadequate reporting of participatory engagement and governance arrangements. In view of the rapid pace of urbanization, we recommend more attention on the potential for urban NbS such as sustainable drainage systems, green roofs and walls, parks and street trees to help with managing flooding and heatwaves while supporting health and wellbeing in cities. In general, we recommend a more systematic approach to gathering and reporting evidence on the process and outcomes of NbS projects in order to build the evidence base and maximize opportunities to learn about what works in different contexts.

This review can support evidence-based deployment of well-designed NbS in relevant government policy in Bangladesh, including plans for climate change adaptation, mitigation, sustainable development, and biodiversity. NbS are context-specific, but many of the lessons learnt in Bangladesh are more widely applicable. By building on the experience and lessons learnt from deployment of NbS over the last few decades, Bangladesh is well placed to lead the way in showing how other countries and communities around the world can protect and enhance their natural assets in order to address multiple societal challenges sustainably.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

AUTHOR CONTRIBUTIONS

AS and TT performed the systematic review and analysis, AS, TT and HI wrote the paper, NS, BT and AC developed the methodology for the systematic review and contributed to the paper.

FUNDING

This study was supported by a Natural Environmental Research Council Knowledge Exchange Fellowship to NS (NE/R002649/1), with additional funding from the Waterloo Foundation.

ACKNOWLEDGMENTS

We are grateful for input from our colleagues Isabel Key and Stephen Woroniecki at the Nature-based Solutions Initiative, who helped us to develop the assessment framework and criteria to assess the strength of the evidence base.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2021.737659/full#supplementary-material

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frontiers in Environmental Science

BRIEF RESEARCH REPORT
published: 07 December 2021
doi: 10.3389/fenvs.2021.678446

Stakeholder Perceptions of Nature-Based Solutions and Their Collaborative Co-Design and Implementation Processes in Rural Mountain Areas—A Case Study From PHUSICOS

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OPEN ACCESS

Edited by:
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Pennan Chinnasamy,
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Specialty section:
This article was submitted to
Interdisciplinary Climate Studies,
a section of the journal
Frontiers in Environmental Science

Received: 09 March 2021
Accepted: 16 November 2021
Published: 07 December 2021

Citation:
Lupp G, Huang JJ, Zingraff-Hamed A, Oen A, Del Sepia N, Martinelli A, Lucchesi M, Wulff Knutsen T, Olsen M, Fjøsne TF, Balaguer E-M, Arauzo I, Solheim A, Kalsnes B and Pauleit S (2021) Stakeholder Perceptions of Nature-Based Solutions and Their Collaborative Co-Design and Implementation Processes in Rural Mountain Areas—A Case Study From PHUSICOS.
Front. Environ. Sci. 9:678446.
doi: 10.3389/fenvs.2021.678446

Planners and engineers increasingly discovered nature as a source of inspiration to mitigate hydro-meteorological risks resulting from extreme weather events. Actors are realizing advantages of such solutions known as Nature-Based Solutions (NBS) to rapidly adapt to changing climate patterns and related impacts such as flooding, landslides, mudflows or rockfalls. NBS also provide multiple co-benefits such as an increased landscape value for society and biodiversity. Because of their inherent characteristics, NBS implementation are more efficient when supported by participative approaches. At the same time, strengthening democratic and collaborative planning into Living Labs approaches generates an increase in interest. This helps to overcome bottlenecks when implementing measures and provide common ground to provide space for new ideas, to promote innovation and to develop solutions with high acceptance. While co-design and implementing NBS has already been applied and well documented for urban areas, there are few publications on collaborative planning, stakeholder perception and NBS co-implementation in rural mountain areas. In our case study analysis from the EU-funded H2020 project PUSICOS, we present stakeholder views on NBS, their possibility to reduce natural hazards in different mountainous case study areas, different discussed measures, NBS types and stages of implementation. We analyze expectations on Living Lab processes to co-design NBS and important topics to be addressed in these processes from the view, perspective and perception of local stakeholders. Despite the importance of NBS on political and research agenda, in both the literature and the interviews, the concept and ideas are less familiar to stakeholders. NBS are mainly encountered within river restoration measures. The main interest was to reduce risks and to find solutions that were attractive and interesting also from an economical point of view e.g. business models for farmers and landowners and less of the multiple benefits that are most important for stakeholders in urban areas. The collaborative planning approach was seen as important for engaging stakeholders and creating knowledge about NBS. These insights will contribute to the understanding and address the management of intense stakeholder involvement processes, identify barriers that arise, and support in-depth participatory processes.

Keywords: Nature-Based Solutions, stakeholders, living Labs, stakeholder perspectives, perception, Acceptance, collaborative planning

1 INTRODUCTION

Climate change causes the increase of extreme hydro-meteorological events triggering floods, landslides, mudflows, avalanches or rockfalls (Kumar et al., 2020). Nature-Based solutions (NBS) are increasingly considered as suitable, viable solutions to increase the effectiveness of technical solutions. For example, they can partially or fully replace static flood protection infrastructures or reduce exposure or vulnerability for landslides, avalanches and rockfall, reduce negative impacts by drought or heatwaves (European Commission—Directorate-General for Research and Innovation, 2015). NBS received a lot of attention in recent years, even reaching the top of both political and research agendas (Nesshöver et al., 2017; Frantzeskaki et al., 2019). The European Union defines NBS as "Solutions that are inspired and supported by nature, which are cost-effective, simultaneously provide environmental, social and economic benefits and help build resilience. Such solutions bring more, and more diverse, nature and natural features and processes into cities, landscapes and seascapes through locally adapted, resource-efficient and systemic interventions. Nature-based solutions must therefore benefit biodiversity and support the delivery of a range of ecosystem services" (European Commission—Directorate-General for Research and Innovation, 2015). The IUCN describes NBS as "Actions to protect, sustainably manage and restore natural or modified ecosystems that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits" (Cohen-Shacham et al., 2016). Examples given by the IUCN are restoring and sustainably managing wetlands, conserving forests, restoring drylands, developing green infrastructure in urban environments and using natural coastal infrastructure with a multitude of benefits for mitigating, and adapting to climate change, enhancing biodiversity but also linking these benefits to the targets of the sustainable development goals such as food security, economic development, education and health benefits (Cohen-Shacham et al., 2016). NBS can be considered as an "umbrella term" for these wider range of concepts and practices (Nesshöver et al., 2017; Pauleit et al., 2017). Benefits of NBS are seen in both providing increased resilience and multiple co-benefits such as increased landscape values for society and biodiversity (Cohen-Shacham et al., 2016; Raymond et al., 2017). The European Union also indicates that NBS can be a source of innovation, with possibilities to deliver multiple benefits across different social groups in a range of environmental, economic and cultural settings and address the challenges of changing climate. The Paris Agreement signatories explicitly refer to NBS to help achieve the mitigation of climate change and as a key adaptation strategy. Mainstreaming NBS is also a core aim in global agendas to deliver on the Sustainable Development Goals (Martin et al., 2021).

While NBS have received a lot of attention in urban areas, this is not the case for rural mountain areas. With their greater exposure to risk and vulnerability to climate change, NBS not only reduce risk to the local population but can also reduce accumulating small events and prevent them from becoming large-scale disasters in densely populated areas downstream (Solheim et al., 2021). PHUSICOS intends to demonstrate the effectiveness of NBS and their ability to reduce the impacts of extreme hydro-meteorological events in rural mountain landscapes. PHUSICOS works on a broad range of NBS addressing hydrometeorological risks across the different case sites. Potential solutions address flooding and water storage/retention, rockfalls, avalanches, landslides, water quality and runoff from agricultural areas, river restoration and novelty NBS to stabilize mountain slopes. An important aspect of PHUSICOS is the upscaling potential of the measures to be implemented.

However, the number of implemented NBS is still low. Unsupportive governance (Kabisch et al., 2016; Ershad Sarabi et al., 2019), various barriers such as lack of political commitment (Solheim et al., 2021) and missing inter-sectorial communication (Zingraff-Hamed et al., 2020a) slow down or halt their implementation. One key success factor to overcoming bottlenecks resulting from a lack of cooperation is in-depth stakeholder involvement right from the beginning. Intense collaborative planning among different public and private actors, as well as citizens for the design and implementation of solutions from the initial stages is recognized as an efficient tool to solve complex problems and to find innovative designs. Recent studies identified that such partnerships and collaborative approaches are crucial for successfully implementing NBS (e.g. Zingraff-Hamed et al., 2021) and creating acceptance, sense of ownership and ultimately, the success of measures and their implementation (Lupp et al., 2021). It is therefore important to involve all relevant stakeholders to ensure a well-functioning co-design process and to deal with potential conflicts, issues, and constraints that may arise (Zingraff-Hamed et al., 2020b). Formalized procedures for collaboration and participation are vital to support the design and implementation of solutions (National Research Council, 2008), and they are increasingly becoming mandatory in projects (Scolobig et al., 2016). Identifying and addressing stakeholder values, interests, and knowledge is a crucial first step for such collaborative processes (Burgers and Farida, 2017). Especially, understanding their skepticism and how to motivate them to act is important to orchestrate collaborative planning (Lupp et al., 2016).

Thus it is important to know how different stakeholders perceive hazards and potential NBS solutions. NBS often require integrated measures which implies collaboration and the willingness of stakeholders to act (Heitz et al., 2009). According to Heitz et al. (2009), risk perception and striving to implement solutions is based on own experiences, beliefs, and psychological, social, economic, temporal or institutional factors. A number of theoretical approaches exist to describe perception of risks, behaviors and actions. Mañez et al. (2016) extend a model of risk perception as a stepping stone for taking actions based on cultural backgrounds, socio-political factors and cognitive affective factors that are influenced by individual and collective backgrounds. Pagliacci et al. (2020) outline the varieties of rationalist and constructivist approaches with the Protection Motivation Theory and Protective Actions Decision Model being the most frequently applied. They are rooted in Planned Behavior theories and consider subjective norms, attitudes, perceived behavioral control and background factors influencing decisions triggering action. Venkataramanan et al. (2020) highlight the willingness to make changes depending on a variety of factors such as awareness of the problem, knowledge, attitudes, intentions that lead to implementing or adopting solutions.

However, stakeholder involvement and perceptions are quite frequently examined from a theoretical point of view in the literature. Actual stakeholder views on NBS are far less frequently explored. For mainstreaming and upscaling of NBS and creating acceptance and a perception among stakeholders that NBS are a suitable and desirable solution, a key aspect is the evolution of the perception and awareness of NBS in such collaborative processes. This aspect is largely missing in literature.

The objective of this paper is to give preliminary insights into the stakeholder perspective of the ongoing collaborative planning and design processes in PHUSICOS. It intends to give insights into stakeholder views and perceptions, awareness and expectations of NBS in the collaborative processes, and the role of engagement and collaborative processes surrounding NBS. It will provide an initial outlook on aspects that are important to raising awareness and improving the perception of NBS as desirable solutions for different stakeholders.

The main research questions are:

- What are the perceptions of NBS or neighboring concepts?
- What are the main interests and concerns of such solutions? Are there differences between urban and rural mountain settings?
- What expectations do actors have regarding collaborative planning of NBS?

The paper presents initial results from the PHUSICOS project and provides insights from the beginning of an intensive in-depth collaborative planning process using Living Lab approaches as systematic, theoretic, and formalized approaches for collaborative planning and co-designing processes (Fohlmeister et al., 2018). With a variety of applications and approaches, Living Labs can be seen as a methodology, system concept, or an environment. The key elements are openness, knowledge development, learning processes for all participants, and meeting on equal ground, including the ones initiating such a process. Other key elements are putting the ones that are affected in the center of the processes and focusing on collaboration with stakeholder involvement right from the beginning to form a quadruple helix innovation network to engage end users (e.g. citizens, NGOs), the public sector (administrations, policymakers), the private sector (businesses) and academia (Lupp et al., 2021).

To provide initial outcomes of the collaborative processes at the different case sites in the coming years, stakeholders will be interviewed continuously with different methodological approaches to assess their perspectives on NBS, learning processes, expectations towards NBS, collaborative planning and co-design, and lessons learned from the collaborative work. For collaborative planning, PHUSICOS applies a Living Lab concept to support and institutionalize intensive collaboration of stakeholders for the co-creation, co-design and co-monitoring of NBS. Despite some fuzziness resulting from a wide variety of activities carried out under the umbrella term “Living Labs”, this concept provides a systematic approach and a framework for stakeholder engagement to provide guidance through different phases of the NBS co-creation processes. A quadruple helix network is in the center of the co-creation process engages stakeholders from the private sector, end users such as citizens and their representatives, the public sector like administrations and academia meeting on equal grounds (Fohlmeister et al., 2018; Lupp et al., 2021).

2 MATERIALS AND METHODS

2.1 Literature Review

With the growing popularity of NBS, the number of terms used to describe or conceptualize them has seen an explosive increase. Therefore, we realize our literature review cannot be exhaustive to include the abundance of all terms and all literature that exist such as reports or presentations that currently exist and opted for the following search strategy (also **Figure 1**).

First, we conducted a literature review between February 10 and 23, 2021 using the Web of Science, Scopus and Google Scholar databases with key search terms that were relevant to stakeholder perceptions of nature-based solutions. It was considered important to use more than one database since search algorithms may vary across databases. In this case, each database yielded a few unique publications which the other databases did not find. We first searched from all three databases with the search terms: (sustainable drainage OR NBS OR nature-based solution OR disaster-risk reduction OR eco-disaster risk reduction OR eco-drr) AND (stakeholder awareness OR stakeholder perception OR stakeholder attitude).

We also used some terms on neighboring concepts of NBS to collect work on stakeholder perspectives from these fields that

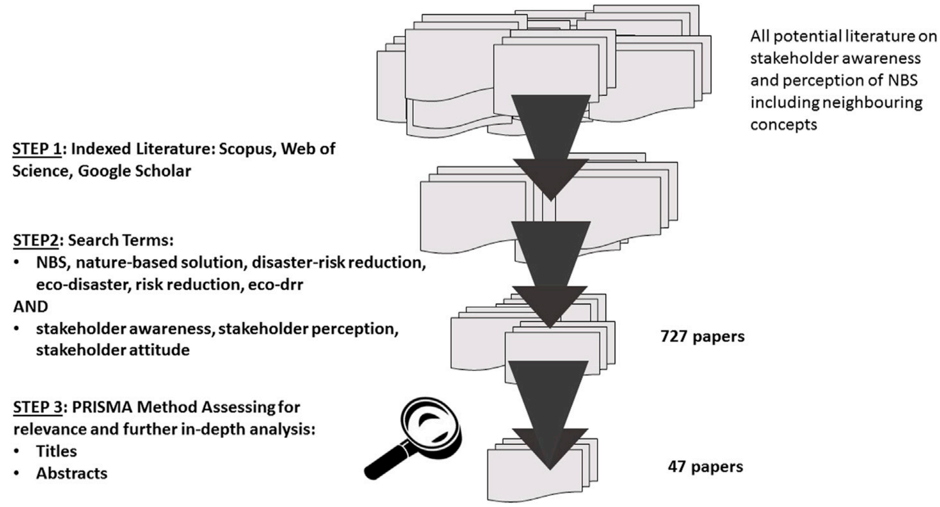

FIGURE 1 | Flow chart on the search strategy and analysis for the literature review.

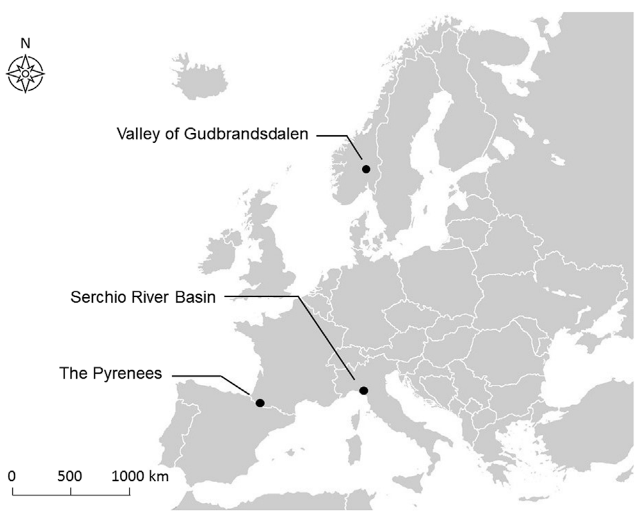

FIGURE 2 | Location of the different PHUSICOS concept case study sites.

promote a similar intention: more natural or nature-inspired solutions to reduce risks, exposure and vulnerability of natural hazards triggered by hydrometeorological events.

A total number of 727 papers were identified. We utilized the PRISMA method (Moher et al., 2009) to identify the most relevant papers. First, we assessed the titles of these papers for relevance and categorized them based on relevance. Then, we assessed the abstracts of the papers with the most relevant titles to further determine which papers would be useful for our research. In this way, we identified 49 relevant publications. We then reviewed the content and extracted the relevant information to be incorporated into our research for a qualitative content analysis (Mayring, 2000).

2.2 Case Study Approach

A case study approach is seen as a flexible way for in-depth investigations at small scales that balance breadth and depth (Taylor, 2016). The studies chosen are the three PHUSICOS demonstrator sites for developing and implementing NBS for reducing the risk from natural hazards in rural mountain areas. The demonstrator case sites are located in the valley of Gudbrandsdalen, Norway, in the Pyrenees, France and Spain, and in the Serchio River Basin, Italy (**Figure 2**).

The ***Valley of Gudbrandsdalen*** is 140 km long and one of the most populated valleys in Norway. It extends from the village of Dombås in the north to Lillehammer in the south. The river valley is extensively used as farmland. Many settlements are located close to the river. The area is exposed to a range of hydro-meteorological hazards such as flooding of the main river and its tributaries, debris flows and debris slides, rockfalls and snow avalanches. Potential measures to address these issues are the reestablishment of floodplains and enhancement of the water storage capacity in the catchment areas (PHUSICOS, 2021).

In the ***Pyrenees***, a mountain range between France, Andorra and Spain, reforestation can help to cope with hydro-meteorological extreme events by reducing the hazard intensity. Afforestation in the release areas can reduce the risk of avalanches. For reducing hazards arising from rockfalls and debris, reshaping a slope through terracing techniques to support with the establishment of vegetation to stabilize the sediments have already been successfully applied a century ago and serve as an inspiration for new measures in the region (PHUSICOS, 2021).

The ***Serchio River Basin*** in Tuscany, Italy is of national interest according to Italian law and has been identified as a “river basin district” for implementation of EU’s Water Framework Directive. A combination of challenges include extreme drought and flooding, seismic risk as well as water pollution by runoff of sediment and nutrients from adjacent farmland. The set of proposed measures include re-vegetation efforts and farming practices to stabilize the soil and to reduce the

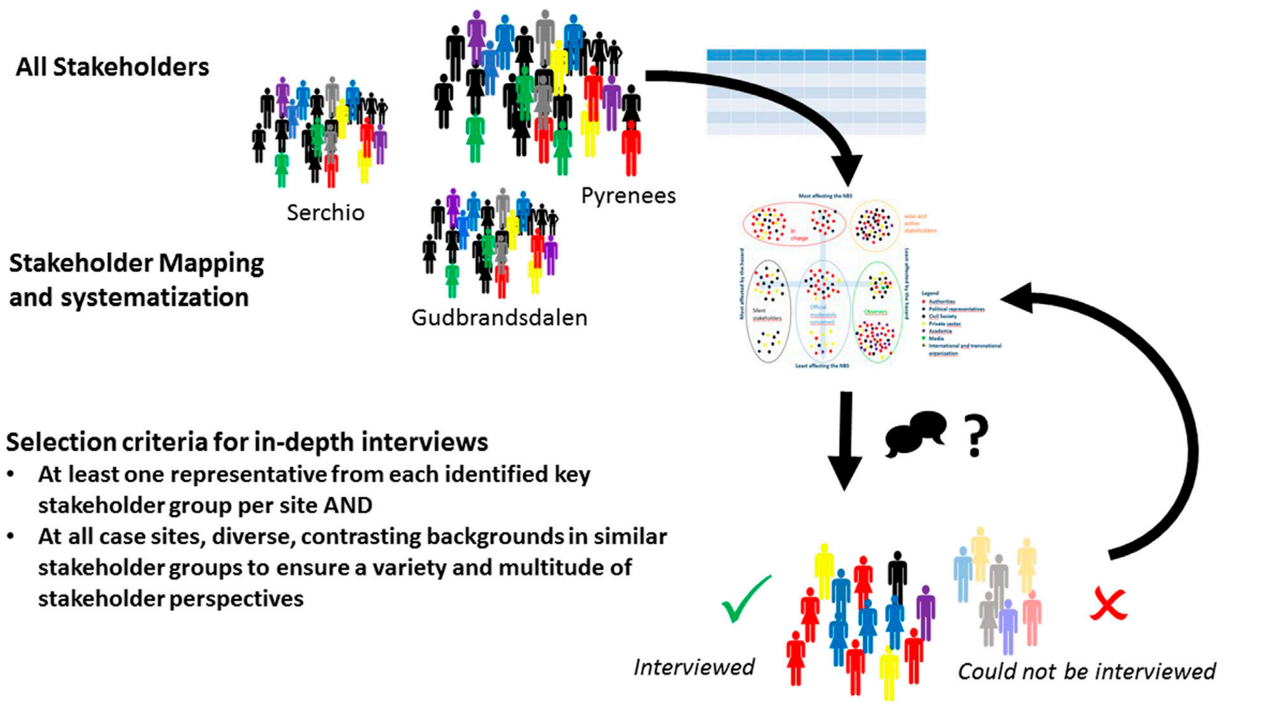

FIGURE 3 | Selection process of interviewees.

runoff from the agricultural fields to the water bodies and Lake Massaciuccoli (PHUSICOS, 2021).

For these demonstrator cases, various stakeholders are actively involved throughout the project using living labs in order to incorporate their knowledge, preferences, views, values and attitudes. One of the main goals of the project is to involve and motivate stakeholders to shape and co-design the implementation of NBS.

2.3 Qualitative In-Depth Interviews

To assess the stakeholder perspectives on NBS, the in-depth participatory processes and in-depth collaborative planning approaches by using a Living Labs approach, we opted for a qualitative approach (Atteslander, 2003). A comparatively small group of interviewees were used in this approach to collect in-depth understanding with semi-structured protocol interviews being developed for this purpose (Marshall and Rossman, 1998).

To cover different perspectives, attitudes and opinions, a systematic approach to select interview partners was chosen according to the principle of maximum contrasts based on the grounded theory (Strauss and Corbin, 1990). The aim was to cover a wide range of perspectives within a small group of interviewees. Criteria could be differences in sociodemographic characteristics, different professional backgrounds and different opinions. Recruiting interview partners followed the following approach (also **Figure 3**).

In considering PHUSICOS's different sites and to allow cross-site comparisons, first, a systematic stakeholder identification task was conducted following an approach developed by the PHUSICOS's sister project RECONECT (Hüesker et al., 2019). Based on systematic stakeholder mapping described by Zingraff-Hamed et al. (2020b), potential stakeholders were listed based on available information from the different sites and on their documentation and available protocols from initial stakeholders meetings within the PHUSICOS project. Based on this information, a list of stakeholders was compiled and assigned to a stakeholder group. The local facilitator teams in charge of the stakeholder processes were asked to add more potentially relevant stakeholders, for example by replacing those not responding to their invitations, unwilling to participate or relevant only for a single or certain steps during later stages of the collaborative planning and co-creation process (Lynam et al., 2007; Reed et al., 2009). Based on the concept of interest-influence matrices and three-dimensional power-influence-attitude grids (Murray-Webster and Simon, 2006), local facilitators were asked to evaluate the roles of stakeholders as well as their importance in the different co-design, co-implementation and co-monitoring/evaluation stages, their relation and affectedness by natural hazards, NBS and decision processes on finding potential solutions to reduce natural hazards.

Based on the results of the stakeholder mapping, interview partners at the different sites were selected for an interview in an iterative process. At each site, at least one representative from the commercial sector, academia, authorities, political representatives and from civil society (represented e.g., by NGOs) would be part of the interview panel. Across all case sites, different backgrounds and sociodemographic features to provide potentially very differing views, perspectives and backgrounds of the interviewees were considered according to Hunziker (2000) to encompass a broad range of perspectives.

However, not all of the initially identified persons (around 20) could be interviewed and other persons had to be chosen instead. Some refused the request for an interview or were unavailable in the given timeframe. Also, some potential interview partners were difficult to reach during the COVID-19 pandemic, and approaches such as collecting interviews in suitable, good environments for building trust for exchange that are important for such qualitative interview approaches (Elwood and Martin, 2000), were difficult to realize. This might have led to a lack of willingness to participate at an early stage of the collaborative processes as well.

Interviews were conducted by phone or video-calls. Notes were taken when interviewees rejected to be recorded. Recorded interviews were transcribed and translated to English for the assessment. The texts were then analyzed, shortened and structured to highlight the key statements and relative frequencies according to Mayring (2000) and Mayring and Brunner (2010).

3 RESULTS AND DISCUSSION

3.1 Key Findings From the Literature Review and Discussion

3.1.1 Nature-Based Solutions, Neighboring Concepts and Stakeholder Perspectives

Despite the broad scope of the literature search including neighboring concepts and disaster risk reduction, not much work on stakeholder perceptions on strategies to reduce risk with NBS or similar concepts could be found. Interestingly, disaster risk reduction, similar concepts and stakeholder perspectives mainly relate to understanding their perception of natural hazards, risks, vulnerability and preparedness to react to an occurring disaster, e.g. evacuation. Related to disaster risk reduction, there is not much mentioned about measures to reduce the risks, or exposure of natural hazards. Buchecker et al. (2013) state in their work that risk perception approaches in literature with a spotlight on disaster risk reduction have a strong theoretical nature and often focus on the perception of risks rather than the perception of risk prevention measures.

Han and Kuhlicke (2019) scanned 1834 NBS papers for stakeholder perspectives and perceptions of NBS in literature but only found 15 papers addressing how people value and perceive the co-benefits of NBS and related concepts. Ferreira et al., 2020 conducted a systematic literature review on NBS with a focus on urban areas related to establishment of green infrastructure (GI) and sibling concepts and came up with 142 papers on stakeholder perspectives. Piacentini and Rossetto (2020) analyzed stakeholders in water related NBS and GI which were almost all situated in urban and peri-urban areas in Mediterranean France and Italy. They found little interest and response from rural areas on these concepts and rather low awareness of concepts of water-related NBS.

3.1.2 Knowledge About Nature-Based Solutions Concepts

Bark et al. (2021) described in their study from the United Kingdom on Natural Flood Management (NFM), that two-thirds of the respondents considered themselves familiar with NFM, however, only 8 strongly considered themselves experts. Understanding and information was collected mainly by participation in one or more NFM projects. In our case studies, we can demonstrate that also other channels are important sources of knowledge of more natural solutions, especially training at universities or through institutions for collecting a basic understanding. In their study, Heitz et al. (2009) describe the farmer's self-conception being "experts for soil", getting information from the Farmers' Trade Union, technical papers and agricultural advisors, but in their examined case, farmers often have a weak awareness of muddy flood risks.

3.1.3 Perceived Positive Features of Nature-Based Solutions

Findings from Han and Kuhlicke (2019) suggest that co-benefits are valued positively and important for many stakeholder groups. This result is confirmed by Pagano et al. (2019), particularly if people have direct access to NBS in urban settings and can interact with them frequently. However, the studies assessed by Pagano et al. (2019) focused only on co-benefits related to recreational and aesthetical aspects and other possible positive aspects such as health, wellbeing, cultural values, and economic development have not yet been considered. Interestingly, our respondents did not emphasize the co-benefits for society so much and more emphasis was placed on benefits for nature, economic opportunities and especially on reducing natural hazards.

3.1.4 Concerns About Nature-Based Solutions

Bark et al. (2021) received a mixed response on evidence of the general effectiveness of NBS and at high flows there was also some concern from the stakeholders we interviewed. In the assessment of stakeholders by Bissonnette et al. (2018), more information was needed on the biodiversity and ecological functionality of NBS. Many participants believed that an economic evaluation of services provided by ecosystems is necessary in order to design effective planning interventions.

Several authors for rural settings claim negatively perceived economic aspects as important concerns or barriers to implement NBS, such as Portugal Del Pino et al. (2020). Piacentini and Rossetto (2020) refer to expected high maintenance costs but that stakeholders consider that additional co-benefits might outweigh the high costs. Bissonnette et al. (2018) stated, that many participants believed that an economic evaluation of services such as recreation or aesthetics is necessary to design effective planning interventions. In the case of adapting to sea level rise in Scotland by Liski et al. (2019), rural stakeholders claimed that decision-making should be based on economic rationality and locally derived evidence and that poorly designed schemes might lead to increased maintenance costs. Willingness to manage flood risks with NBS was accepted only if there would be evidence that considerable numbers of residents would benefit from them with increased protection.

TABLE 1 | Interviewed stakeholders (anonymized).

Stakeholder	Stakeholder group according to Zingraff-Hamed et al. (2020b)
Agriculture 1 (business)	Commercial sector
Agriculture 2 (family-owned)	Commercial sector
Research, agronomist	Academia
Water administration (region)	Authorities
Water administration (county)	Authorities
Authority (region)	Authorities
Authority infrastructure 1 (province)	Authorities
Authority infrastructure 2 (province)	Authorities
Nature manager community	Political representatives
Forest administration	Political representatives
Decision maker county	Political representatives
Decision maker community	Political representatives
Representative of interest group for nature and outdoor recreation	Civil society

Pagano et al. (2019) refer to 10% of those interviewed explicitly preferring traditional grey solutions as they are well known and reliable. According to the authors, this stresses again the importance of demonstration pilots and capacity building. The PHUSICOS stakeholders also consider this to be an important aspect and expectation that PHUSICOS as a project could provide such opportunities.

3.1.5 Important Aspects of Nature-Based Solutions From Stakeholder Perspectives

While co-benefits, usability and "neat looking" solutions are very important in urban areas (Hoyle et al., 2017) for gaining acceptance, they might contradict the foundational purposes of NBS such as enhancing biodiversity, and addressing natural hazards, which should be a stronger focus for rural and mountainous areas. Besides validating the durability of NBS, the importance of the economic aspects of NBS that were highlighted in our interviews also agrees with the literature where NBS are implemented in more rural settings. This is also in line with some managerial perspectives from urban areas. Bark et al. (2021) stated that for stakeholders, solutions should be cost-effective and without issues of tenure and coordination of such solutions. Heitz et al. (2009) also highlights economic issues playing a role in mitigation measures for mudflows. Pagano et al. (2019) found that 24% of their respondents perceived the construction and maintenance costs and efforts as limitations to the diffusion of these systems. Santoro et al. (2019) also highlighted that stakeholders expressed the need to have quantitative assessments of the effectiveness of the selected measures in reducing flood risk and expected impacts with specific reference to the costs and benefits of the chosen actions.

3.1.6 Stakeholder Involvement and Participatory Approaches

While Wamsler et al. (2020) critically reflected on stakeholder involvement with limited individual personal interests and a lack of environmental awareness in urban contexts, in the study by Buchecker et al. (2013), the interviewees experienced the participatory process as an effective means of sustainable decision-making. Only one person who was interviewed doubted that the broad involvement of stakeholders would result in a feasible solution. Others, who had been initially skeptical in this respect, changed their minds during the process.

3.2 Interview Findings

A total of 13 persons agreed to participate in the interviews covering all stakeholder groups from different levels except the two groups media and international organizations (**Table 1**), which usually are observers rather than intensively involved stakeholders in the co-creation processes (Zingraff-Hamed et al., 2020b).

While the concept of NBS has received a lot of attention among both research and international and European policy in urban areas, the attention received in rural mountain areas is to a lesser extent. Around one third of the interviewees first encountered NBS and their related terminologies with PHUSICOS. The others encountered it with river restoration measures, related to agricultural practices, forestry and one interviewee encountered the NBS concept in an urban context. In the majority of the cases, the information on NBS was received from universities.

Expectations of the Innovation Action funded by the European Commission were related to several aspects (**Supplementary Figure S1**). Most stakeholders wanted to get new ideas on how to address natural hazards with new solutions and on the project to serve as a starting point to reduce the risks in their area. An important aspect for all stakeholder groups was the desire to find solutions that are attractive and interesting from an economical point of view (e.g., a new business model for farmers and landowners).

> "*(NBS) offer opportunities that envisage viable alternative measures*" (Agriculture 1).

With the pan-European perspective of the project including a retrospective learning case, upscaling and replication of good NBS solutions were perceived to be an attractive opportunity provided by the project. In the words of an interviewee from an authority at the regional level:

> "*The PHUSICOS Living Labs project generates a positive impact on the territory and on the bodies that manage it in terms of dissemination and information of cutting-edge green engineering techniques*" (Authority regional level).

Also, expectations from civil society and NGOs as representatives on the process and the Living Lab approach:

> "*I think PHUSICOS has great potential for inspiration. (Our institution) wants more people to adopt a mindset and thinking around green solutions. It can contribute to decision-makers adopting more sensible solutions. (. . .) If PHUSICOS can help lift the focus, something (our*

institution) and its members have been lobbying for over many years, this would be a welcome addition" (Representative of interest group for nature and outdoor recreation).

At the beginning of the Living Lab process, NBS were mainly seen as beneficial for nature and providing interesting opportunities for local businesses.

"*These solutions are renewable, they have a fairly small carbon footprint and it's better for boosting the local economy.*" (Forest administration political representative).

Only to a lesser extent were other benefits mentioned, such as risk reduction, higher acceptance by the public or multiple benefits. The main concerns were a lack of profitability or a lack of local value added by NBS, the perception of being less reliable and skepticism from many stakeholders towards NBS.

"*The costs can be higher; there is a lack of thinking about the maintenance of these solutions which could be very dangerous in the long term. I am quite skeptical about some of these solutions*" (Forest administration political representative).

Barriers to implement NBS were seen in a multitude of issues. One set of perceived barriers was the validation of the effectiveness of NBS or applicability at the case site and the time needed for NBS to work (especially related to vegetation). Other barriers were related to human factors such as a lack of knowledge of NBS stakeholder acceptance or a lack of collaboration (**Supplementary Figure S2**). Or, in the words of an interviewee:

"*Compared to the grey solutions that have a parameterized dimensioning, the solutions based on nature, in general, suffer from defined dimensioning parameters that allow to determine their goodness and therefore their applicability to extrapolated situations. The human aspects would be especially associated with the lack of companies as well as of available and contrasted know-how to be able to undertake the action successfully.*" (Authority infrastructure 2).

When asked for their expectations of Living Labs, most interviewees mentioned the aspects of engaging stakeholders and creating knowledge regarding NBS. Most expressed interests related to economic aspects of NBS; such as

"*Combining protection issues with the survival of private businesses which today are the real guardians of the territory*" (Agriculture 1).

Other points were raising awareness and stakeholder engagement and the desire to see that NBS is demonstrated to be effective for their region as the main outcome of the Living Lab process.

"*By sitting people together, one will achieve a common ground and a common understanding of the problems and challenges. (. . .) A good physical measure that also safeguards the natural values in a sustainable way. Increased understanding among all stakeholders,* "*make them see the light*"!" (Decision Maker County).

Other expected goals to be achieved with Living Labs is to successfully disseminate NBS solutions, raise awareness and provide learning opportunities.

3.3 Comparing and Discussion the Literature Findings and Interviews

Comparing the findings from the literature with the interviews from PHUSICOS (**Table 2**), we can underline a lack of knowledge of ecosystem-based, near-natural or nature-based solutions. This highlights the huge importance of opportunities for learning surrounding NBS and its benefits. This could also take place both in an indirect way, such as visiting implementation sites and discussing implemented projects or in a direct way such as providing documentations, brochures, and newsletters on the topic. In many cases described in the literature, the effectiveness of NBS are perceived very critically and with much skepticism. PHUSICOS stakeholders that were interviewed provided a more positive perspective. Nonetheless, they considered more learning and demonstration of the durability and effectiveness of NBS in particular to be useful.

"*Best practice is very important. People are interested in how solutions have been implemented in other places where there is flood risk. If you can show solutions that are working elsewhere, people will take note. There is a lot of learning in good examples.*" (Authority Region).

Knowledge institutions such as academia are another key group that can be involved in the learning process.

Academia can play an important part to generate more awareness and positive perception on NBS. They are able to provide a basic understanding of NBS and are widely accepted by most of the other stakeholders as a neutral actor.

Experts might present various challenges, and give examples of green solutions, and then you could discuss these (Decision Maker County).

Stakeholder mapping showed that academia is the major component of both stakeholder groups, "the wise and active stakeholders" and "the observers" (Zingraff-Hamed et al., 2020b). Both groups are mainly in the least affected by the hazard and/or the least affecting the NBS implementation category so they are often not a core actor of the collaborative planning and therefore, in a neutral position.

In addition to demonstrating the durability of NBS, the literature focused on urban areas emphasizes the importance of co-benefits for society. In rural mountainous areas, peculiar

TABLE 2 | : Comparison between literature findings and interviews with stakeholders.

Description	Findings from the literature review	PHUSICOS stakeholders' answers
Stakeholder familiarity with NBS and related concepts	Lack of knowledge, but land users and farmers consider themselves the experts [e.g.Heitz et al. (2009)]. Most of the literature underlines the importance of NBS projects for learning and raising awareness/knowledge [e.g. Bustillos Ardaya et al. (2017); Pagliacci et al. (2020)]	About one third have not encountered the concept of NBS before the start of PHUSICOS, "entry-point" knowledge often provided from universities and related contacts
NBS benefits perceived by stakeholders	Mainly urban NBS in the literature, mainly co-benefits for society are valued Han and Kuhlicke (2019), managerial views relate to easier maintenance Bark et al. (2021)	Interviewees mainly refer to benefits for nature and express potential economic opportunities
Concerns of stakeholders on NBS	NBS are less effective especially in severe events Pagano et al. (2019), high maintenance costs Portugal Del Pino et al. (2020), little acceptance for solutions that are not aesthetically pleasing Hoyle et al. (2017)	Evidence of durability or functionality is largely missing, effectiveness is lower, maintenance is more costly, fear of invasive species
Perceived barriers to NBS by stakeholders	Often, a lack of knowledge and awareness of evolution and importance of participation [e.g. Venkataramanan et al. (2020); Buchecker et al. (2013)]	Lack of knowledge, PHUSICOS project approach could help to overcome or address this issue
Collaborative processes	Mixed experiences, critical reflections [e.g. Wamsler et al. (2020)] as well as positive reports Buchecker et al. (2013)	Expectations relate to raising awareness, learning, experiencing hands-on cases, collecting experiences, demonstrating effectiveness and viability, and new attractive business models

technical, environmental and socio-economic features can be detected (Baills et al., 2021; Strout et al., 2021), and these environments deal more often with natural hazards such as flooding, landslides, rockfalls or snow avalanches (Evans and Clague, 1994; Haritashya et al., 2006). Implemented NBS in the catchment areas can provide benefits and risk reduction for the entire watershed including heavily populated urban areas (Albrecht et al., 2017).

highlighted importance of the economic aspects of NBS in our interviews with the stakeholders can be found both in literature where NBS are implemented in more rural settings and in literature presenting managerial perspectives of NBS in urban areas. Economic aspects and financing of NBS are identified as one of the major challenges to mainstream NBS. A number EU research- and innovation-funded projects currently strive to increase the knowledge on funding and business models for NBS (Mayor et al., 2021). A key challenge is that benefits provided or created by nature and NBS are largely public goods and services that either have no markets or market prices or have costs that are relevant only in the distant future (Schweppe-Kraft and Grunewald, 2015).

Several attempts have been undertaken to describe the value of nature, both in monetary and non-monetary terms also as a basis for the development of business models. Calculating economic values is already considered challenging. Only a few studies on the monetary value of NBS in comparison with grey solutions exist (Debele et al., 2019). This is due to the fact that the monetary value of the social co-benefits is difficult to assess and it is even more difficult to adequately consider them in collaborative processes (Perosa et al., 2021).

The difficulty of monetizing especially socio-ecological benefits raises additional challenges for private businesses such as farmers. Neither are they included in market prices, nor are they are deeply rooted in relevant mechanisms such as the EU Common Agricultural Policy with funding regimes and subsidies heavily influencing the business models of farmers–despite the efforts taken for integrating strong greening and environmental requests (e.g. Lupp et al., 2014 and Lupp et al., 2015 for the case of biomass production for energy purposes). Thus, private stakeholders are hesitant in engaging or investing financially in NBS, and most business models currently depend on direct or indirect interventions to mitigate the lack of market mechanisms through policy instruments and incentives valuing the benefits generated by NBS (Mayor et al., 2021).

Nonetheless, the power of economic instruments has been demonstrated by the results of The Economics of Ecosystems and Biodiversity (TEEB) study (Kumar, 2011), and some of the created co-benefits can be conveyed in economic value. Examples of this include increased real estate value next to urban green or restored river sections in urban areas (Luttik, 2000; Gerwien, 2020) or payment schemes for ecosystem services such as the financial support of the Munich Public Works for organic farming in the Mangfall Valley. This included marketing activities of dairy products originating from the area in order to avoid excessive, expensive technical purification for their main source of drinking water for the city (Grolleau and McCann, 2012).

Efforts therefore strive to demonstrate for stakeholders the cost-benefit within broader assessment approaches. These approaches aim to describe the benefits of NBS by accounting both monetary and non-monetary values through combining multidisciplinary approaches and integrating physical, environmental, social, human and economic features (Dumitru and Wendling, 2021). Multi-criteria tools comprising a set of indicators based on the application of multi-criteria decision analyses including stakeholders in both the weighting and assessment procedures can help to compare the effectiveness of different nature-based, hybrid and grey design solutions (Pugliese et al., 2020).

Finally, despite several attempts to explain stakeholder perception and awareness, explanation models do not elucidate why stakeholders take action or not (Lindell and Perry, 2012). Appraisal of risks and knowledge does not necessarily result in protective action and may even cause dissonant attitudes. Even with much awareness on NBS, the motivation to take action is often low, or grey engineering solutions are preferred. Key motivational factors to generate action can be sharing of the responsibility of implementing a solution among different stakeholders, the collective engagement in prevention, and the importance of dialogue (Maidl et al., 2019).

4 CONCLUDING REMARKS AND OUTLOOK

With the findings from the literature review and with the first interview round of selected PHUSICOS stakeholders, a number of challenges need to be addressed to overcome the lack of a broader implementation and mainstreaming of NBS.

Despite their importance on political and research agendas, the knowledge of on-the-ground stakeholders on NBS is limited. Sometimes, there is even a lack of awareness about the natural hazards and exposure to the risks. In many cases described in the literature, the effectiveness of NBS is perceived very critically and much skepticism exists towards such solutions. While the PHUSICOS stakeholders that were interviewed provided a more positive perspective towards NBS, still for this group, learning opportunities, site visits and "hands-on cases" to demonstrate the durability and effectiveness of NBS was seen as particularly useful. Knowledge institutions such as academia can be an important group to contribute to such a learning process and play an important role as trustworthy, neutral actors in the NBS negotiation processes. Applying concepts such as Living Labs with its philosophy of all stakeholders contributing by meeting on equal ground can create an atmosphere of trust and understanding to support finding and implementing NBS solutions.

While for public landowners or real estate developers in urban areas, the creation of multiple co-benefits might already be a critical driver for successfully implementing NBS, the situation in rural mountain areas can be more complex. Landowners providing the space for NBS solutions and in particular farmers often initially perceive NBS as a limitation to their economic potentials for gaining revenues from their productive land. It is therefore vital for this group to demonstrate with real-life examples that NBS can be an interesting opportunity also in economic terms. Co-benefits of NBS therefore need to be measured and valued in a clear, widely accepted manner with sound methods, tools and indicators. Assessment frameworks can describe value arising for the different stakeholders and can lay the foundation for business models related to the implementation, managing, monitoring of NBS or resulting new opportunities from such measures.

With many expectations expressed at the beginning by the interviewed stakeholders in the PHUSICOS project, it will be interesting to follow up with the stakeholders though the Living Lab processes and the evolution of their perceptions on NBS. Learning, building trust and intensive in-depth collaboration processes might be the key elements for triggering action in real life, mainstreaming NBS and gaining equal or more acceptance of NBS over traditional grey solutions.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author.

ETHICS STATEMENT

In line with the Research Ethics Procedures of the Technical University of Munich and project partner institutions based on the mentioned EU, EEA and respective country regulations, the participants received written information on how the data would be used and were asked to give their consent to participate in the interviews according to these guidelines. We obtained consent from all research participants prior to the interviews and handled their confidentiality and interview data according to this consent.

AUTHOR CONTRIBUTIONS

Conceptualization: GL, AZ-H; research design, methodology: GL, AZ-H, JH; validation, SP; AO and AS; literature review: GL and JH; investigation at sites, conducting interviews, transcripts, translation: NS, AM, TW-K, MO, TF, EM-B, and IA; writing: GL, JH, and AZ-H; project administration: AS, AO, BK, and ML; funding acquisition, AO; BK, supervision, AO, AS, SP, and ML.

FUNDING

PHUSICOS has received funding from the European Union's Horizon 2020 research and innovation program under grant agreement No. 776681.

ACKNOWLEDGMENTS

The authors like to thank the site owners, key actors and stakeholders of the different case sites for their willingness, availability and spending their time to participate in the in-depth interviews and sharing their views and perspectives.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2021.678446/full#supplementary-material

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Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

frontiers in Environmental Science

ORIGINAL RESEARCH
published: 07 February 2022
doi: 10.3389/fenvs.2021.690011

To Plant or Not to Plant: When can Planting Facilitate Mangrove Restoration?

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OPEN ACCESS

Edited by:
Boris van Zanten,
World Bank Group, United States

Reviewed by:
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Nehru Prabakaran,
Wildlife Institute of India, India
Pelayo Menendez Fernandez,
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Specialty section:
This article was submitted to Interdisciplinary Climate Studies, a section of the journal Frontiers in Environmental Science

Received: *01 April 2021*
Accepted: *30 December 2021*
Published: *07 February 2022*

Citation:
van Bijsterveldt CEJ, Debrot AO, Bouma TJ, Maulana MB, Pribadi R, Schop J, Tonneijck FH and van Wesenbeeck BK (2022) To Plant or Not to Plant: When can Planting Facilitate Mangrove Restoration? Front. Environ. Sci. 9:690011. doi: 10.3389/fenvs.2021.690011

Global change processes such as sea level rise and the increasing frequency of severe storms threaten many coastlines around the world and trigger the need for interventions to make these often densely-populated areas safer. Mangroves could be implemented in Nature-Based Flood Defense, provided that we know how to conserve and restore these ecosystems at those locations where they are most needed. In this study, we investigate how best to restore mangroves along an aquaculture coast that is subject to land-subsidence, comparing two common mangrove restoration methods: 1) mangrove restoration by planting and 2) Ecological Mangrove Restoration (EMR); the assistance of natural mangrove regeneration through mangrove habitat restoration. Satellite data revealed that historically, landward mangrove expansion into the active pond zone has mainly occurred through mangrove planting on pond bunds. However, there is potential to create greenbelts along waterways by means of EMR measures, as propagule trap data from the field revealed that propagules of pioneer species were up to 21 times more abundant in creeks of the pond zone than near their source in the coastal zone. This was especially true during the prevailing onshore winds of the wet-season, suggesting that smart seasonal sluice gate management could help to efficiently trap seeds in target ponds. In the coastal zone, field experiments showed that permeable brushwood dams, aimed at expanding mangrove habitat, could not sufficiently overcome subsidence rates to increase natural mangrove expansion in the seaward direction, but did significantly increase the survival of already established (planted) seedlings compared to more wave-exposed sites. The survival and growth rate of EMR-supported plantings greatly varied between species. Out of the four planted species, *Rhizophora mucronata* had the highest survival (67%) but the lowest growth rate. Whereas the pioneer species *Avicennia alba* and *Avicennia marina* had lower survival rates (resp. 35 and 21%), but significantly higher growth rates, even resulting in fruiting young trees within a 16-month timeframe. Overall, we conclude that 1) EMR has potential in the pond zone, given that propagules were observed to reach well into the backwaters; and 2) that mangrove recovery in the coastal zone may be facilitated even at very challenging coastal sites by combining EMR with the planting of pioneer species.

Keywords: mangrove restoration, EMR, planting, aquaculture, land subsidence, building with nature

1 INTRODUCTION

Climate change effects such as sea level rise and an increased frequency and severity of storms threaten many coastlines around the world (IPCC, 2019). The impact of global climate change is often magnified by regional problems such as land subsidence (Hallegatte et al., 2013; Nicholls et al., 2021), and land use change (Zhang et al., 2021), leading to coastal encroachment. Hence, there is a pressing need for intervention strategies with which to adapt to climate change and reduce flood risks along populated coastlines. Coastal wetlands such as mangroves and salt marshes have gained interest as important ecosystems to reduce vulnerability of coastal communities (e.g. Temmerman et al., 2013; Zhu et al., 2020; Zhang et al., 2021). Coastal wetlands are able to attenuate waves even during storms (Möller et al., 2014; Willemsen et al., 2020), dampen storm surges (Stark et al., 2015; Montgomery et al., 2019), trap sediment and grow with rising sea level (Kirwan et al., 2016). This makes conservation and restoration of such wetlands interesting for coastal protection, and consequently economically interesting. Not only in areas where construction of conventional coastal defense structures, such as sea walls and levees, is not feasible (Winterwerp et al., 2013), but also to reduce costs of such conventional structures at locations where they are feasible (Sutton-Grier et al., 2015; Narayan et al., 2016; van Wesenbeeck et al., 2016; Schoonees et al., 2019).

In tropical regions, mangrove restoration is regarded as a widely applicable strategy to enhance coastal safety (e.g. Alongi, 2008; McIvor et al., 2012). In this, planting has long been a favored approach (Ellison, 2000), especially in the lower intertidal zone where land rights are often not an issue (Erftemeijer and Lewis, 1999). However, this type of planting projects have often given low success rates due to a combination of inappropriate species selection (non-pioneer species) (Ellison, 2000; Primavera and Esteban, 2008) and/or unfortunate site selection (mudflats below mean sea level where mangroves would not naturally occur). When planting is reported to be successful in terms of survival, there is the risk of hampered natural succession, as seedlings are often planted at such high density that the resulting stand leaves little sunlight for potential natural recruitment (Barnuevo et al., 2017; Pranchai et al., 2018; Proisy et al., 2018).

To overcome plantation failure, interests have been shifting increasingly towards restoration of mangrove habitat to promote natural recruitment, rather than using active planting of mangrove propagules or seedlings (Balke and Friess, 2016; Lewis, 2005; Winterwerp et al., 2020, 2013). The focus of mangrove habitat restoration (i.e. EMR), lies in mitigating the establishment thresholds that limit natural propagule settlement and survival (Lewis and Brown, 2014). Lewis (2005) demonstrated the importance of addressing the first threshold, (i.e. limited propagule availability) (**Figure 1**, panel 0), by digging a creek, thus reestablishing aquatic connectivity at a propagule-deprived location and thereby initiating mangrove regeneration over the course of 6.5 years. At more exposed sites, propagules need to overcome additional thresholds to establish successfully for which they need to surpass three size-dependent windows of opportunity: 1) a flooding-free phase, 2) followed by a wave-free period and 3) an erosion-free period (**Figure 1**, panels 1–3) (Balke

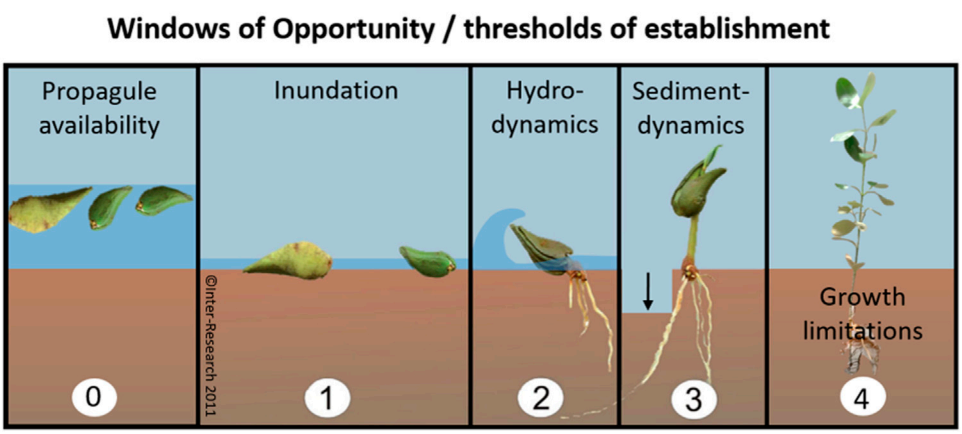

FIGURE 1 | Windows of opportunity that propagules should encounter (or threshold of establishment that propagules should overcome) before they can colonize a site. EMR makes use of measures that extend the window of opportunity for natural establishment **(0–3)**. Mangrove planting is either aimed at overcoming propagule limitation **(0)** or at skipping the thresholds of propagule establishment **(1–3)** by using larger seedlings or saplings that would, to a certain extent, be able to withstand conditions that are too harsh for smaller seedlings. Finally, saplings need to overcome limitations of growth **(4)** such as predation, disease or other stressors to grow into reproductive young trees (**panels 0 and 4** are additions to Balke et al. (2011)'s Windows of Opportunity figure).

et al., 2015, 2013, 2011). Van Cuong et al. (2015) demonstrated that these windows of opportunity can be extended along eroding shores by placing permeable fences parallel to the coastline. These fences both increased the bed level through sediment trapping (i.e. creating first window) and mitigated the wave stress (i.e. creating second window), allowing natural seedling recruitment to increase from zero to 24,000 seedlings ha^{-1} over the course of 3 years. The examples provided by Lewis and Van Cuong et al. clearly show that informed mitigation of stressors restricting natural mangrove recruitment can be sufficient for successful restoration. It is noted that this approach requires proper understanding of the natural system, as the optimal method may differ depending on the specific abiotic and biotic constraints at the selected sites.

Despite the shift in interest towards EMR instead of mangrove planting, the latter remains a popular practice. From a scientific perspective, planting of larger seedlings may be expected to be effective, as it can either overcome propagule limitations (**Figure 1**, panel 0) or skip many of the size-dependent establishment thresholds that propagules need to surpass (**Figure 1**, panel 4). After all, if propagules have established and grown taller, they can persist at locations where newly established seedlings would be uprooted (Balke et al., 2015, 2013, 2011). In that sense, there may be merit to mangrove planting when the purpose is to accelerate mangrove recovery by skipping the most sensitive propagule life-stages at sites where EMR is not entirely sufficient to improve all conditions required for natural colonization. In other words, could mangrove planting help to accelerate the recovery process when it is combined with EMR, especially at challenging locations?

The purpose of this study was to advance current insights into the best mangrove restoration practices, using a 20 km coastline stretch of Demak district, Java, Indonesia as example of a challenging restoration site. The area has a history of aquaculture and is marked by the associated impaired hydrological connectivity. The ongoing subsidence and subsequent erosion of the shoreline further complicate mangrove restoration. Various restoration projects, initiated by the government, NGO's and local communities, have however attempted to expand the district's existing mangroves into a greenbelt using either planting or EMR measures. In seaward direction, restoration has mainly been attempted through planting of *Rhizophora* species. Although more recently, sediment-trapping brushwood dams have also been implemented as an EMR measure to facilitate natural mangrove expansion (Tonneijck et al., 2015). In landward direction, mangrove restoration has mainly focused on compensating the loss of terrestrial vegetation, brought on by salinization after the conversion of rice paddies to aquaculture in the past. Local communities and NGO's therefore planted *Rhizophora* spp. in the pond zone, both to re-enforce the pond bunds and to create a local source of firewood. More recently, EMR in the active pond zone has also been initiated in the form of the creation of mangrove habitat in active ponds lining the waterway. To this end, pond owners have partitioned their pond, and sacrificed the parts lining the creek for mangrove rehabilitation, motivated by higher yields from mangrove-associated aquaculture (Bosma et al., 2020). The observed natural expansion in the coastal zone, the various attempts to plant mangroves, and the EMR measures in both the coastal zone and aquaculture pond zone have all had their positive effect on the mangrove cover in Demak in the last decade.

We here evaluated both the effectiveness of the already installed EMR-measures and mangrove plantings, and the effectiveness of combined EMR-measures and plantings in an experimental setting. The aim was to investigate how mangrove regeneration can best be achieved in landward and seaward direction from the subsiding coastline. We explored this question with the following sub-questions: 1) how has mangrove expansion occurred in seaward and landward direction in the past, mainly through planting or mainly through natural expansion? 2) Can hydrological EMR measures (e.g. sluice gate management) induce natural mangrove recovery in the pond zone (i.e. would enough propagules be available at landward sites if hydrological connectivity to target ponds was increased)? 3) Can wave-reducing and sediment-trapping measures (EMR-dams) induce natural mangrove recovery at challenging sites in the coastal zone by increasing the chances of a) new seedling establishment or b) survival of established seedlings? and; 4) Can mangrove planting in combination with EMR-dams accelerate mangrove recovery at challenging coastal sites?

2 METHODS

2.1 Site Description

The study site is located along the coast of Demak district at 6.53°S, 110.30°E on Java, Indonesia. The shore is characterized by a mixed, mainly diurnal tide, with a tidal range of 1.1 m (MMAF, 2012). The region has an average annual rainfall of 2,200 mm (Suryadi et al., 2018), with a dry season dominated by an offshore wind (SE) from June to August, and a wet season with onshore wind (NW) from December to February (MMAF, 2012). During the wet season, the maximum significant wave height 4 km offshore is reported to be 1.5 m, with a period of 5.5 s (Tonneijck et al., 2015). The onshore waves during this season leave the coastline of Demak prone to erosion and flooding, which are further exacerbated by land subsidence. These processes have led to two major erosion events of Demak's shoreline: One between 2002 and 2003 causing the coastline directly east of Semarang city to retreat with 3 km, and one between 2007 and 2009 affecting our study area, 5 km east of Semarang (**Figure 2**). This erosion event removed the majority of the aquaculture ponds 1 km seaward of the coastal road, making the coastal road an unintended coastal defense structure.

2.2 Rationale

We studied the natural and assisted processes of mangrove expansion in landward direction (i.e. the pond zone) and seaward direction (i.e. the coastal zone) from the current coastline in the project area (the old coastal road). The pond zone in the study area is characterized by active traditional aquaculture ponds, and drowning abandoned aquaculture

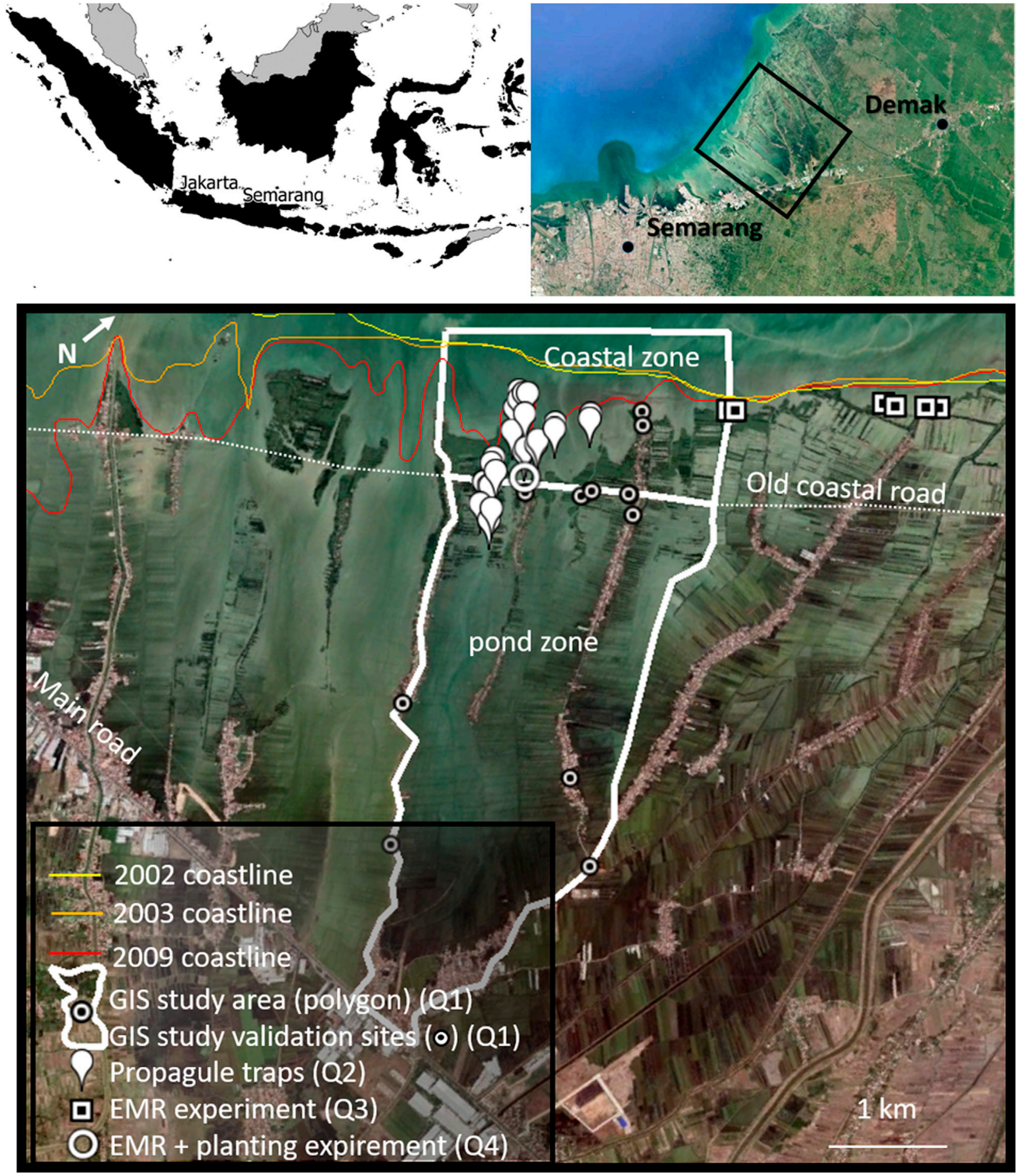

FIGURE 2 | The focus area of this paper, the coastline of Sayung subdistrict (black rectangle) in Demak, Indonesia. Historic coastlines are indicated (before, in between and after two major erosion events), as well as the old coastal road which became the last standing structure resisting coastal erosion, dividing the active aquaculture zone from the dynamic coastal zone. The locations of the four experiments linked to the four research questions are indicated with the marks displayed in the legend. Detailed location descriptions and overviews per experiment are shown in the method section.

ponds towards the coast. The coastal zone in the study is defined as the area ranging from MHW to MLW (the maximum depth at which the EMR-dams are placed), which ranges roughly 0–600 m from the shoreline (Tonneijck et al., 2015). In order to shed light on best practices for mangrove recovery, we addressed research questions one to four in the introduction using the following correspondingly numbered methods. 1) GIS monitoring of natural and planted mangrove vegetation in the coastal zone and pond zone using time series of satellite images (GIS study in **Figure 2**). 2) A year-round propagule monitoring campaign seawards and landwards of the old coastal road (propagule traps in **Figure 2**) to assess the potential for natural mangrove colonization in the active pond zone if maximal EMR-hydrology would be applied. 3) A field experiment at multiple coastal sites with different wave exposure conditions to understand the effect of EMR-dams on mangrove recruitment and seedling survival (EMR-experiment in **Figure 2**). Finally, 4) a field experiment to study the effect of mangrove planting in combination with EMR-dams on seedling growth and survival (EMR + planting in **Figure 2**). These four experiments are further elaborated on below.

2.3 Q1. GIS Study: Which Mangrove Recovery Process is Dominant in Each Zone?

To quantify the importance of the two contrasting processes of mangrove recovery in Demak, we quantified changes in vegetation types (i.e. natural mangroves, planted mangroves and terrestrial vegetation) in the coastal zone and the pond zone of the study area (**Figure 2**) over multiple years. We analysed high resolution (<1 m^2) satellite images in Google Earth for vegetation cover, starting in 2005 (i.e. before the 2007 erosion event), in 2010, and then on a yearly basis from 2013 until 2018 (**Supplementary Table S1**). The timeline tool of Google Earth displays mosaics of multiple high-resolution satellite images (i.e. acquired at different dates or from different sources) for the area of interest. We therefore attempted to select the most complete image (no mosaic) for each year of interest (**Supplementary Table S1**). Due to limited coverage, we were obligated to use a mosaic of two different years for the baseline year 2005, where the majority of the image was comprised of a 22-4-2005 image, and a small corner of the area of interest was dated 31-5-2003. Vegetation cover in both 2003 and 2005 was very low however, so the mosaic was deemed appropriate to use as a pre-erosion baseline.

Images from each year were exported from Google Earth as kml files and imported to ArcGIS with the “kml to layer tool”. Vegetation was manually digitized per image in the two areas of interest, the coastal zone seawards of the old coastal road, and the pond zone landwards of that road (e.g. **Supplementary Figure S1**). The sparse and fragmented mangroves in the pond zone, interspaced with muddy and algae rich aquaculture ponds, made semi-automated recognition of vegetation cover challenging for our study area. Especially since the imagery was limited to wavelengths in the visible range of the light spectrum (Red, Green and Blue). However, the high-resolution imagery did allow for precise “manual” photointerpretation. A method recognized as reliable for both coastal object identification (Chinnasamy and Parikh, 2020) and fine-scale forest-cover mapping in general (Castilla et al., 2008). In our study area, we identified and categorized vegetation “objects” based on location and pattern. For instance, vegetation along the southern main road (**Figure 2**) or near active rice paddies (i.e. fresh water available) was classified as “terrestrial”, whereas vegetation lining the coast or aquaculture ponds was classified as “mangroves”. Planted mangroves were distinguished from natural stands based on their darker green color and structured pattern of evenly spaced straight lines, as opposed to the grey-green color and more diffuse pattern displayed by natural mangrove stands. Ultimately, we classified all vegetation into one of three categories: natural mangrove cover (predominantly *Avicennia* species), planted mangrove cover (predominantly *Rhizophora mucronata*), or terrestrial tree cover (e.g. garden plants, fruit trees).

Validation of these three vegetation types was conducted during a field-truthing visit by the corresponding author in November 2018 in which the vegetation in randomly selected plots of differing sizes (40–2,400 m^2) was identified to species level (**Supplementary Figure S1**). Within each plot, we also collected forest structure data, both by counting the number of individuals per species and by recording each tree’s diameter at breast height (DBH). From these parameters, we calculated forest parameters such as tree density (n ha^{-1}) and basal area (m^2 ha^{-1}). In addition to validation plots inside the study area, 13 additional validation plots were selected in the pond area 10 km to the northeast, which was still unaffected by erosion and salinization. These served as a reference for terrestrial vegetation diversity in inside the area back in 2005, before the onset of coastal erosion. Field validation of the 2018 GIS vegetation categories showed that the visual characterization of vegetation from the satellite images was accurate, with an overall accuracy of 89% and a kappa-coefficient of 0.81.

Trends in vegetation cover changes over time for planted and natural mangroves were investigated per zone and species with a generalized regression model, using the surface area of the vegetation category of interest as a response variable, assuming a Gaussian distribution for the two mangrove types (planted vs. natural). A logarithmic regression model was used to analyze the decline in terrestrial vegetation. Data from the field campaign in terms of DBH, basal area and stem density were compared between planted and natural mangrove stands with Kruskal-Wallis tests, and the tree density differences between species and stand types was tested with a two-way ANOVA after log-transformation of tree density.

2.4 Q2. Propagule Traps: Can EMR-Hydrology Support Natural Mangrove Recovery in the Pond Zone?

To investigate if natural mangrove recovery in the pond zone could be supported through EMR-hydrology, we studied landwards propagule dispersal throughout the main creek of the pond zone and compared that to propagule abundance in the coastal zone (the propagule source). We did this by trapping floating propagules in modified crab-traps, to compare weekly propagule abundance in both zones. The propagule traps (**Supplementary Figure S2A**) were designed as netted bamboo cylinders 120 cm long, with a circular 55 cm diameter opening at one end. During deployment, 50% of this opening was always above the water surface, as the traps were maintained horizontally in the water using floats. The floating traps were loosely anchored to an anchoring pole in such a way that the trap could rotate freely around the pole, so that the circular opening was always facing the current.

The traps were placed across the creek and throughout the coastal bay. In total, 40 traps were placed throughout the basin, nine traps in the creek of the pond zone and 31 traps in the coastal zone (**Figure 2**, close-up in **Supplementary Figure S2B**) and they were emptied weekly throughout the year in 2017. Data from the wet season are partially missing as the coastal traps were continuously destroyed by waves, despite repeated efforts to replace them. In addition, the rough seas made it hard to reach and empty the remaining traps regularly.

The variability of propagule abundance over time and in space was compared between the pond zone and coastal zone (source zone) using a generalized linear regression model, assuming a negative binomial distribution (package glmmTMB in RStudio version 1.0.143). Propagule counts of the three most abundant species observed in the traps (*Avicennia marina*, *Avicennia alba* and *Rhizophora mucronata*) were used as response variables, using week number and zone of interest (near the source or in the pond area) as explanatory variables. Two other species were sporadically found in the traps, but their numbers were too low for meaningful comparison between the two zones. The effect of week number on the residuals of the model was furthermore tested by fitting a smoothing term using a general additive model (GAM) from R package mgcv to investigate the effects and significance of seasonality (ie. the non-linear effect of "week number") on propagule counts.

2.5 Q3. EMR-Dams: Can EMR-Dams Induce Natural Mangrove Recovery at Coastal Sites?

We quantified the effect of EMR-dams (i.e. sediment trapping brushwood dams) on natural seedling establishment and survival of established seedlings. To do so, we set up a seedling monitoring study using 2 × 2 m plots at coastal sites above mean sea level (MSL) (**Figure 2**). To evaluate the wave sheltering effect of the dams, we set up seven monitoring blocks: three blocks that were artificially sheltered by permeable dams constructed the previous season (EMR-dam sites) (**Supplementary Figure S3C**), three blocks at a naturally sheltered site behind a vegetated chenier (i.e. elevated sand bank with vegetation on top; positive controls) (**Supplementary Figure S3B**), and one block at an exposed site above MSL (negative control) (**Supplementary Figure S3C**). We were only able to include one block at an exposed site, due to the limited availability of wave-exposed sites above MSL. Each block consisted of three 2 × 2 m plots which were used to monitor natural mangrove recruitment in terms of number of seedlings per species, seedling survival and new seedling recruitment over 1 year in 2017 (**Supplementary Figure S3D**).

To investigate the survivability of established mangrove seedlings at these locations, we designed a small-scale planting experiment in which seedlings of two mangrove species were planted in two of the three 2 × 2 m plots per block in the following year (i.e. 2018; **Supplementary Figure S3E**). One of the 2 × 2 m plots per block was used to plant *R. mucronata* seedlings (the species most commonly used for mangrove planting in this area) and the other was used to plant *A. alba* seedlings (the most abundant, naturally expanding mangrove species in the area). The seedlings were planted in a matrix of 5 × 5 individuals per plot, resulting in a planted seedling density of 6.5 m^{-2}. Seedling height and viability was recorded at (t_0), and again after half a year and a full year. Differences in survival between the species and between the shelter-types was tested with a Kaplan-Meijer survival test, and differences in average growth rate between species were tested with a *t*-test.

2.6 Q4. EMR-Dams + Planting: Can Mangrove Planting Combined with EMR-Dams Accelerate Recovery?

To further investigate if and how mangrove planting could accelerate mangrove recovery at sites with EMR-dams, we designed a larger mangrove planting experiment with which we closely monitored the survival and growth of the multiple mangrove species planted at the oldest EMR site in the area (constructed and maintained since 2013, open circle in **Figure 2**). The planting experiment was set up in 20 plots of 2 × 2 m in which four mangrove species were randomly planted in a matrix of 5 × 4 individuals (**Supplementary Figure S4**). The four species that were planted all occurred naturally in the vicinity: *A. alba*, *A. marina*, *R. mucronata* and *Rhizophora apiculata*. Prior to planting, the seedlings were raised in a nursery. *A. marina* and *A. alba* were collected as propagules from the water, and *R. mucronata* and *R. apiculata* were collected as young wildlings from underneath a variety of mature trees in the region to ensure some genetic heterogeneity. Propagules and wildlings were then raised in a nursery for 2 months, of which 6 weeks in the shade and 2 weeks in full sunlight to acclimatize the seedlings to life in the field.

3 RESULTS

3.1 Q1. GIS Study: Which Mangrove Recovery Process is Dominant per Zone?

3.1.1. Satellite Data

The GIS time series of different types of vegetation cover showed a decline of terrestrial vegetation and an expansion of mangrove vegetation throughout both the coastal and pond zone from 2005 until 2018 (**Figure 3**). The 64.9 ha of terrestrial vegetation in the pond zone in 2005 decreased logarithmically with $e^{-0.24}$ $ha.y^{-1}$ ($R^2 = 0.9$, $p < 0.001$) between 2005 and 2018, leaving only 2.8 ha of terrestrial vegetation in 2018. Meanwhile, natural mangrove recovery was especially dominant in the coastal zone and

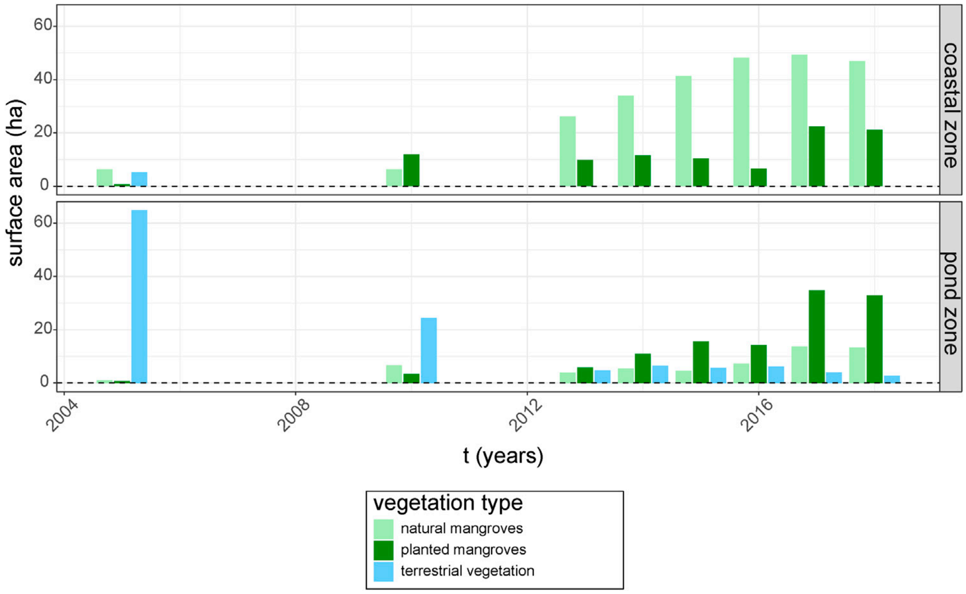

FIGURE 3 | Trends in total surface area (ha), digitized from high resolution images available in Google Earth, of the three vegetation types [terrestrial forestation, planted mangroves (*R. mucronata*) and naturally-recovered mangroves (*Avicennia* spp.)] in the coastal zone and the pond zone from 2005 until 2018.

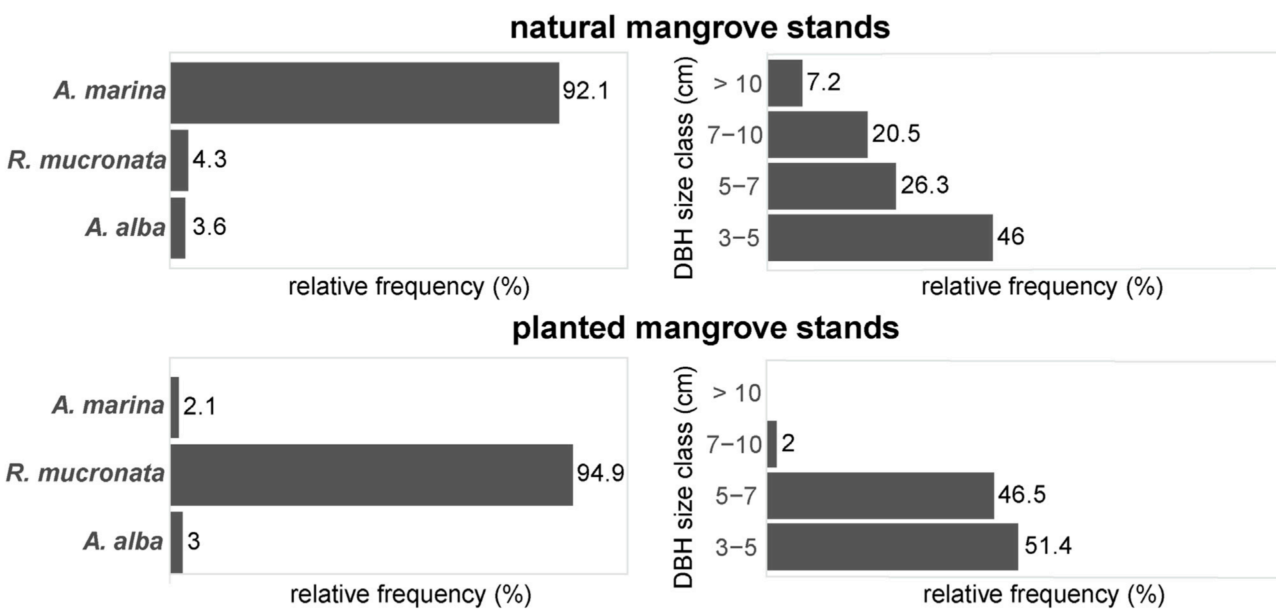

FIGURE 4 | Forest structure of natural and planted mangrove stands in field validation plots of the GIS study in terms of relative tree density per species (left) and the relative distribution of trees per DBH size class (right).

showed an average increase of 3.9 ha per year (R^2 = 0.85, $p <$ 0.001), increasing from 6.3 ha in 2005 to 46.8 ha in 2018. Mangrove expansion through planting was significantly less rapid in the coastal zone ($p < 0.01$), with an average expansion rate of 1.3 ha per year (R^2 = 0.56, p = 0.03).

In contrast, mangrove expansion in the pond zone could be mostly attributed to planting efforts, with an average increase of 2.5 ha per year (R^2 = 0.62, p = 0.01) of planted mangroves as opposed to an average expansion rate of 0.8 ha per year in natural mangrove stands (R^2 = 0.53, p = 0.02). This resulted in 32.9 ha of

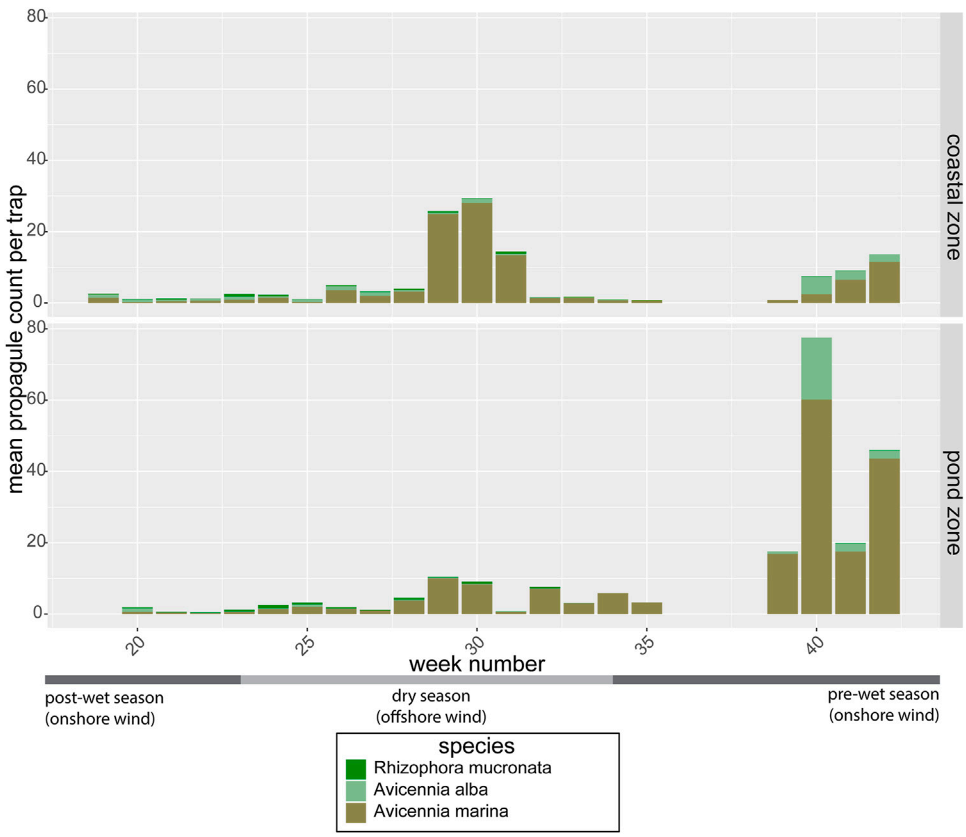

FIGURE 5 | Average weekly propagule counts of the three most abundant mangrove species per trap in the coastal zone (propagule source) and in the pond zone. Data from the mid-wet season are missing due to storm damage to the traps.

planted mangroves in the pond zone in 2018 (almost exclusively on pond bunds), while natural mangrove cover (inside abandoned ponds) amounted to only 13.3 ha. The relative success of planting in the pond zone can most likely be attributed to the much slower natural mangrove expansion in this area ($p < 0.01$), as the absolute expansion rates through planting were not significantly faster in the pond zone compared to the coastal zone ($p = 0.17$).

3.1.2 Field Data

In general, the composition and structure of the mangrove stands classified as "planted" in the GIS study were significantly different from the stands classified as "natural". Planted mangrove stands showed a significantly different species composition than natural mangrove stands ($F = 19.6$, df = 2, $p < 0.001$), with 95% of the planted stands consisting of *R. mucronata*. In contrast, natural mangrove stands consisted of 92% *A. marina*, and only 4% *R. mucronata* (**Figure 4**). There were no significant differences found between planted stands and natural stands in terms of basal area ($m^2\ ha^{-1}$) and tree density ($n\ ha^{-1}$) (**Supplementary Table S2**), although the trees in the natural stands had a significantly larger DBH than the trees in the planted stands ($X^2 = 10.5$, df = 1, $p < 0.01$). Most trees in the planted stands had a DBH of 3–7 cm, whereas the trees in the natural stands were much more diverse in terms of stem diameter, with many young trees, and progressively fewer older and thicker trees (**Figure 4**).

The field observations further revealed that the class of declining terrestrial vegetation was composed of fresh to brackish-water village "forest gardens", used by the villagers for basic fruit and materials provision. Based on a tally of 3.5 thousand trees in remaining forest garden fragments (also including the nearby reference area which was not severely affected by increased salinization), these forests were found to be composed of more than 50 different species of trees, almost all of which had an important utilitarian role in terms of food, forage or materials provision. More details on the composition and uses of these forest gardens as well as their degradation and loss caused by coastal erosion and salinization will be presented elsewhere.

3.2 Q2. Propagule Traps: Can EMR-Hydrology Induce Natural Mangrove Recovery in the Pond Zone?

The propagule traps that were deployed throughout the area proved effective in trapping propagules of multiple mangrove species. Propagules of *A. marina*, *A. alba* and *R. mucronata* were the most abundant (**Figure 5**), although a few propagules of *Rhizophora stylosa* and *Avicennia officinalis* were sporadically

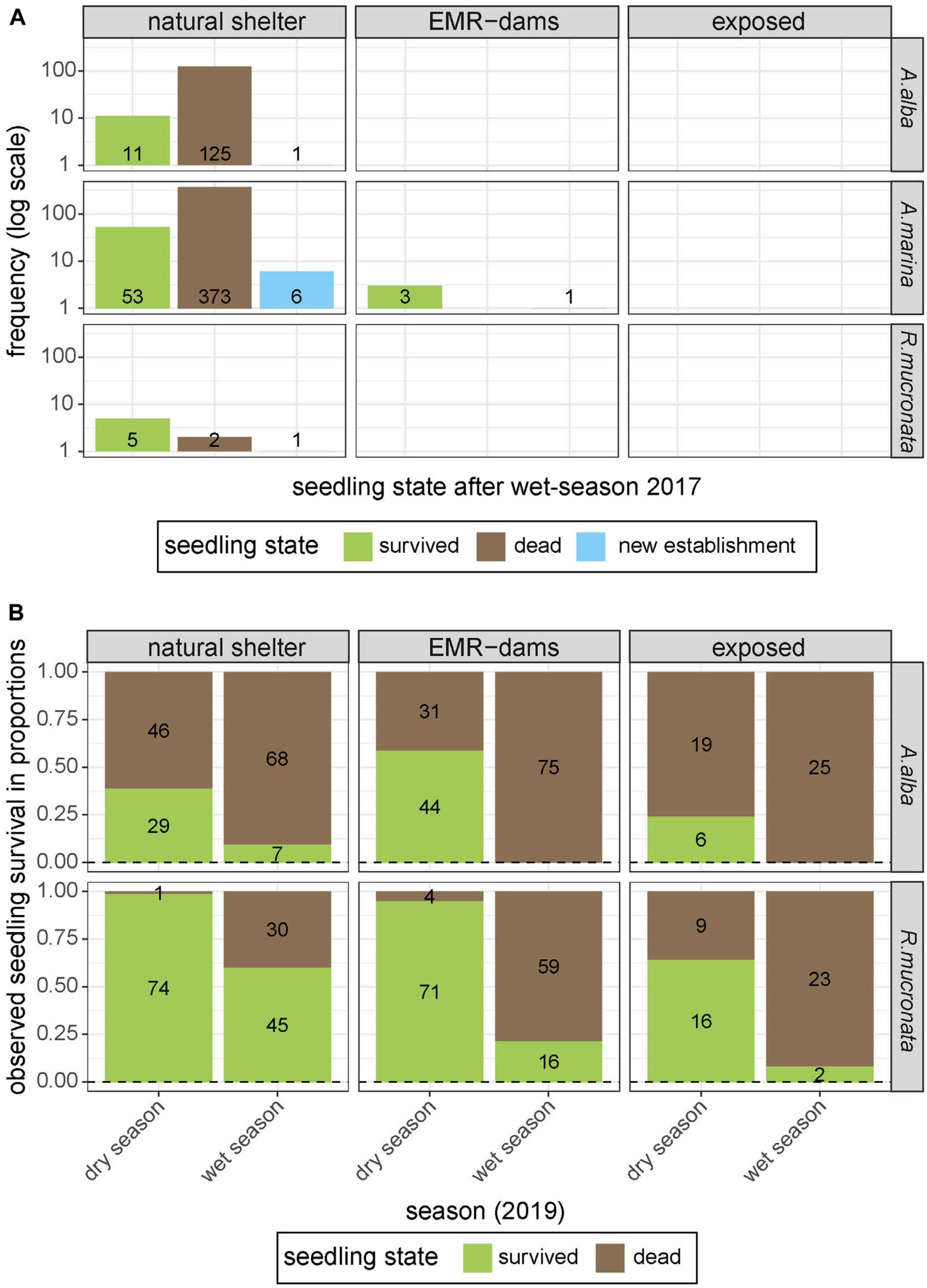

FIGURE 6 | (A) Existing and new natural establishment of mangrove seedlings at the three differently sheltered coastal sites above MSL [natural shelter, artificial shelter (EMR), and wave-exposed], and survival of existing natural recruits over the course of the experiment. **(B)** Survival of *A. alba* and *R. mucronata* seedlings that were planted at naturally sheltered sites, at habitat restoration sites behind an EMR-dam, and at an exposed site.

found as well between week 32 and 35 (not visible in **Figure 5**). The relative abundance of propagules in the coastal zone versus the pond zone depended significantly on the time of year (p = 0.01), with propagule counts in the pre-wet season (week 39–42) being 2 to 21 times higher in the pond zone than in the coastal zone (**Figure 5**). While in the mid-dry season (week 28–31), propagule counts in the pond zone were lower than in the coastal zone. This temporal variation in propagule abundance was caused by the timing of the propagule rains of the two most abundant pioneer species in the area: *A. alba* and *A. marina*.

Both *Avicennia* species showed a clear temporal signal that explained 9.6 and 3% of the variance in seed abundance of, respectively, *A. marina* (gam: $F = 5.6$, edf = 8.2, $p < 0.0001$) and *A. alba* (gam: $F = 3.9$, edf = 2.7, $p = 0.007$). *A. marina* showed a clear peak around week 30 (in the middle of the dry season) and an increase in propagule counts from week 40 onward at the start of the wet season. The pre-wet season peak was visible in both the coastal zone and the pond zone, although more pronounced in the pond zone (**Figure 5**). The mid-dry season peak was only observed in the coastal zone. *A. alba* only showed a peak before the wet season around week 40 and at the end of the wet season around week 20. This could be caused by one big propagule rain over the whole wet season, but due to missing data over the wet season we cannot be sure of this. *R. mucronata* did not show a significant seasonal pattern in propagule availability, with propagules present year round, though in low abundance.

3.3 Q3. EMR-Dams: Can EMR-Dams Induce Natural Mangrove Recovery at Coastal Sites?

Monitoring of plots at coastal sites revealed that most sites behind EMR-dams and the exposed sites could not (yet) support natural mangrove recruitment in 2017. Only the naturally sheltered site harbored an abundance of natural seedlings at the start of the experiment, and showed significant new recruitment of natural seedlings over the course of the monitoring campaign (**Figure 6A**). One of the EMR-sites did have a few seedlings of *A. marina* that survived over the subsequent wet season (**Figure 6A**), and it showed the establishment of one new *A. marina* seedling, suggesting that this particular EMR site had favourable establishment conditions.

The small-scale experimental planting of *R. mucronata* and *A. alba* seedlings in the following year, to quantify the effect of EMR-dams on seedling survival and growth of established seedlings, showed that survival of both species decreased over time (**Figure 6B**). This is an expected natural process in such a dynamic environment, but there were some clear differences between species. Overall, the relatively large *R. mucronata* seedlings showed significantly higher survival rates than the smaller *A. alba* seedlings (Kaplan-Meier, X^2 = 109, p < 0.0001). However, *A. alba* grew 61 mm.month^{-1} faster on average than *R. mucronata* (p < 0.001). Survival rates at the exposed sites and EMR-dam sites were significantly lower than at the naturally-sheltered sites (Kaplan-Meier, X^2 = 27.8, p < 0.0001). However, the EMR-dams did increase the survival of *R. mucronata* compared to the wave-exposed site (binom.test, p < 0.001). Ultimately, the best surviving seedlings were those at the naturally-sheltered site behind a vegetated chenier (Kaplan-Meier, X^2 = 148, p < 0.0001).

3.4 Q4. EMR-Dams + Planting: Can Mangrove Planting Combined with EMR-Dams Accelerate Recovery?

The four species that were planted above mean sea level at an EMR-dam site, but just below the threshold of natural mangrove recruitment, showed clear differences in survival and growth (**Figure 7A**). Over the course of 16 months, the survival rate of the *R. mucronata* seedlings was significantly higher than of the *A. alba*, *A. marina* and *R. apiculata* seedlings (Kaplan Meier, X^2 = 131, p < 0.0001). At the end of the experiment, only 6% of *R. apiculata* seedlings had survived, as opposed to 21 and 35% of the *A. marina* and *A. alba* seedlings, and 67% of the *R. mucronata* seedlings (6% > 21% > 35% > 67%, p < 0.001). However, a comparison of the growth rates of the surviving seedlings (**Figure 7B**) showed that *A. alba* and *A. marina* had significantly higher growth rates than *R. mucronata* and *R. apiculata* survivors (GLM, p < 0.001), which even resulted in two fruiting *A. marina* individuals by the end of the experiment.

4 DISCUSSION

In this study, we aimed to evaluate the potential for- and the effectiveness of EMR and mangrove planting in the pond zone and coastal zone of a site in Northern Java. Our aim was to understand under what conditions planting, EMR or a combination of both can best be applied.

4.1 Mangrove Restoration in the Pond Zone

4.1.1 Q1.a Which Mangrove Recovery Process is Dominant in the Pond Zone?

In the pond zone, GIS information showed that disappearing terrestrial trees were partly substituted by salt tolerant mangroves over time, mostly as a result of planting on pond bunds. The spatial continuity of these planted stands was therefore fragmented, with a very limited patch size (max 5 m wide cross-shore and 200 m long long-shore) of only a single species. Small forest patches are known to be vulnerable to disturbance and likely have little value in terms of coastal protection (Koch et al., 2009) or biodiversity (Hanski, 2015). However, when viewed as fragments to the larger forest, these smaller patches may provide a propagule source for further forest expansion, although natural expansion of *Rhizophora* species was neither observed on satellite images nor in the field, suggesting that there was little suitable natural habitat for this particular species. Nevertheless, the planted stands may account for some ecological connectivity and structural heterogeneity of the larger system (Fahrig, 2017). In addition, the planted patches might still be useful for shade, timber harvest (Van Oudenhoven et al., 2015), mitigation of some of the CO_2 release from the pond bunds (Sidik and Lovelock, 2013), and some limited fish pond water-quality regulation (Rönnbäck and Primavera, 2000). However,

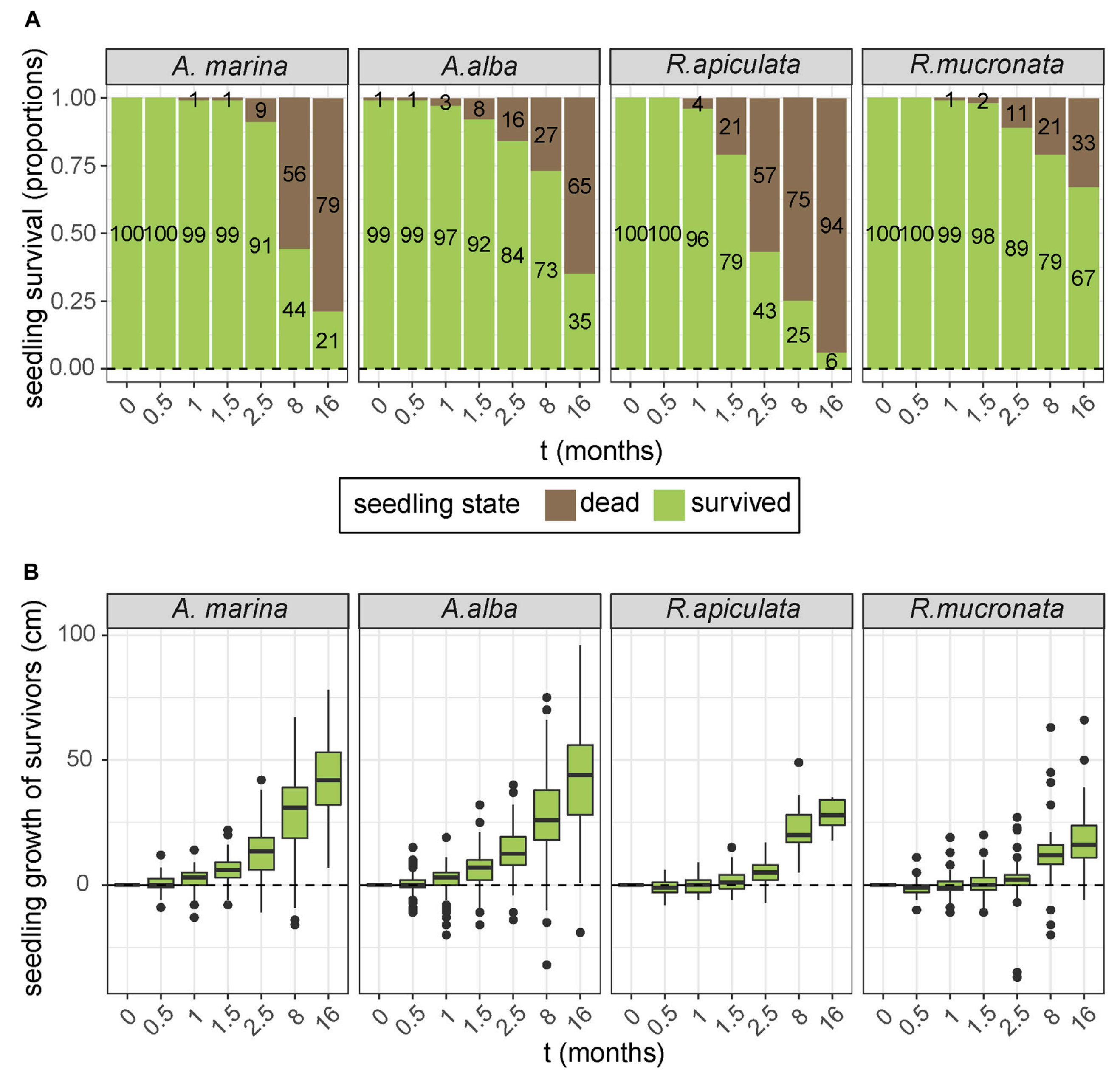

FIGURE 7 | (A) Observed survival over time of four different mangrove species that were planted at a sheltered site behind an EMR-dam around mean sea level. **(B)** Average seedling growth of surviving seedlings from the four planted mangrove species.

decomposing leaves may also worsen the water quality (Bosma et al., 2020). Finally, planted mangrove stands are valued by the local communities for their aesthetic value in a lowland previously devoid of tree-like vegetation (pers. comm. Local communities). While mangrove planting on banks of active ponds may have some advantages, planting mangroves inside disused ponds without EMR measures is reportedly a poor practice, either resulting in slow growth (Proisy et al., 2018) or mortality of planted seedlings (Brown et al., 2014). Also, in the case that natural colonizers can reach the site, additional planting often hampers the growth and succession of natural colonizers (Matsui et al., 2010; Proisy et al., 2018). Therefore, we conclude that planting may be a valuable measure in the pond zone but only when applied on pond bunds when aquaculture ponds are still active, bearing in mind that this will not result in a fully functional mangrove forest (**Figure 8**).

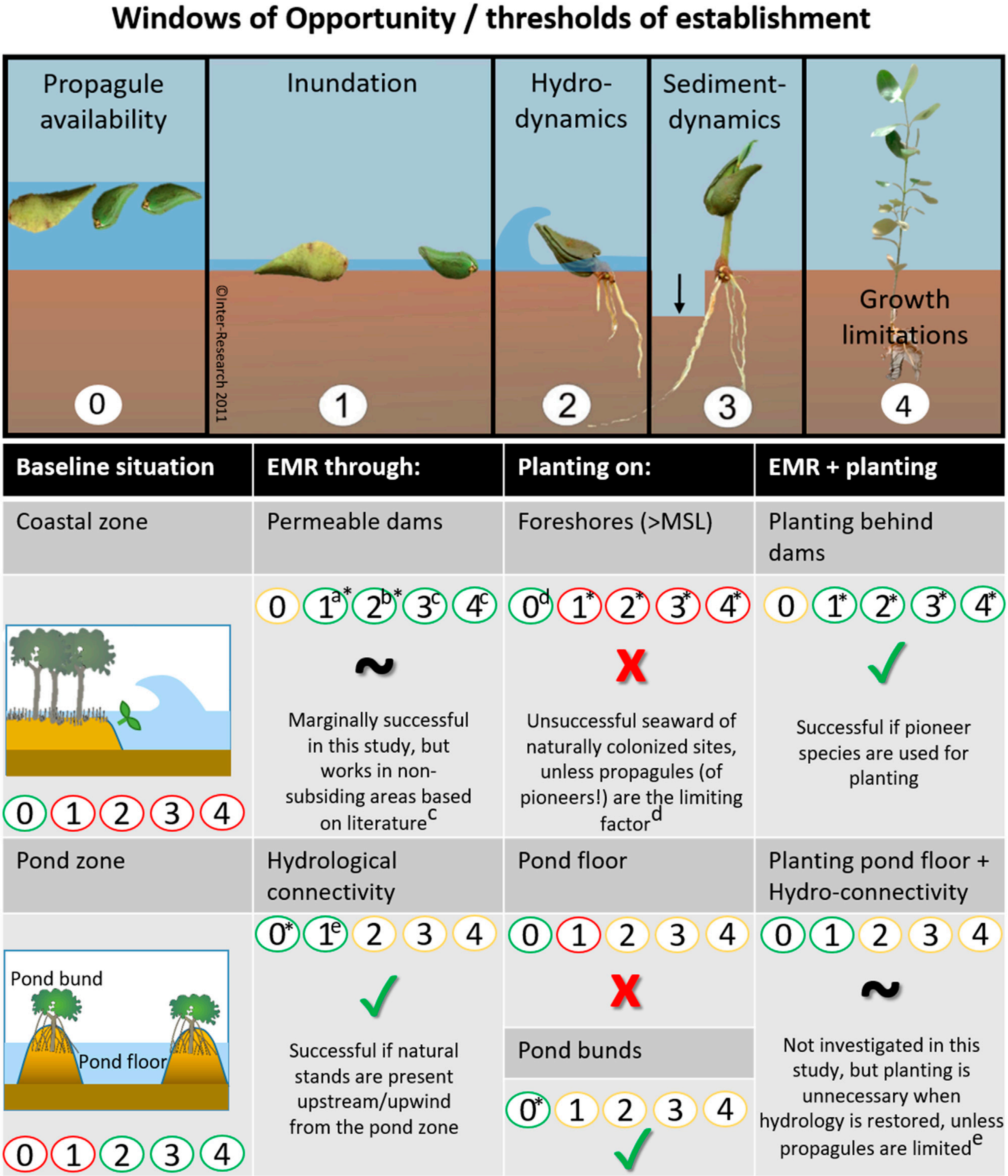

FIGURE 8 | Summary of the effectiveness of different restoration methods (EMR, planting, and a combination of both) to overcome thresholds of establishment in the coastal zone and the pond zone (red: ineffective, green: effective, yellow: condition already met in baseline situation), based on the findings of this study (*) and literature [a: (Cado et al., 2021), b: (Winterwerp et al., 2020), c: (Van Cuong et al., 2015), d: (Saenger and Siddiqi, 1993), e: e.g. (Lewis, 2005; Matsui et al., 2010; Proisy et al., 2018; van Bijsterveldt et al., 2020)]. The top panel of this figure was expanded from Balke et al. (2011)'s original Windows of Opportunity figure (panel 1-3).

FIGURE 9 | Top: Bird's-eye view of the breakwater seaward of Wonerejo, Demak in 2017 (drone image courtesy of CoREM-UNDIP). The white arrow indicates the direction of the picture time series. The time series show the development of a mudflat after construction of the breakwater in 2012, followed by natural colonization in 2013 and subsequent planting of *R. mucronata* in 2014. Note the relative height difference in all pictures of the naturally colonized *A. alba* trees (black arrows) versus the surrounding planted *R. mucronata* trees (2015–2018).

4.1.2 Q2. Can EMR-Hydrology Support Natural Mangrove Recovery in the Pond Zone?

Besides planting on pond bunds, there is also potential for natural mangrove recovery inside aquaculture ponds. That hydrological EMR measures can be effective in ponds has been demonstrated in several studies on the effect of strategic pond breaching in disused or degraded ponds (e.g. Brown et al., 2014; Matsui et al., 2010). In such areas, natural recruitment often fails, either because propagules do not reach the targeted aquaculture pond (Di Nitto et al., 2013), or because the sediment condition of the pond floor is not yet favorable for seedling survival. This was for instance the case in degraded ponds in North Sulawesi, where six planting attempts over the course of 9 years all resulted in total seedling mortality. It was not until man-made drainage channels were filled-in and pond bunds were breached in strategic directions that the sediment condition improved (Brown et al., 2014). These measures subsequently resulted in the recruitment of 32 mangrove species, with overall seedling densities up to 20.000 ha^{-1}. In Demak, disused aquaculture ponds are generally abandoned because pond bunds can no longer be maintained under the rising water levels. Along the coast, these ponds typically fill up with sediment quickly and mangrove recruitment follows not long after (van Bijsterveldt et al., 2020). Recruitment in abandoned ponds therefore does not seem to be an issue in this area. However, our findings show that propagules can travel further inland than where the current natural recruitment occurs based on the satellite data. In addition, despite the lack of mid-wet season data, it is clear that the major propagule release peak of the two pioneer species (i.e. *A. marina* and *A. alba*) co-occurs (at least partly) with the NW monsoon. Wind is known to be an important factor in propagule transport (Di Nitto et al., 2013; Van der Stocken et al., 2015, 2013). Therefore, the onshore winds during the wet season most likely propelled large quantities of propagules further inland, explaining the relative larger abundance of propagules in the pond zone compared to the coastal zone during that season. These observations suggest that there is additional unutilized potential for EMR measures in the active aquaculture zone (**Figure 8**). For instance, season optimized sluice gate management could ensure that sufficient propagules enter target sites in the pond zone, if seasonality of propagule release and wind direction are taken into account. Additional measures, such as pond partitioning and localized pond-floor raising (Bosma et al., 2020), would of course be necessary to create habitat for the propagules to land, and eventually realize mixed-mangrove aquaculture or greenbelt-rimmed ponds along rivers and creeks.

4.2 Mangrove Restoration in the Coastal Zone

4.2.1 Q1.b Which Mangrove Recovery Process is Dominant in the Coastal Zone?

Our GIS study revealed that mangrove expansion in the coastal zone could be attributed mostly to dispersion of pioneer species, and only for a small extent to planting efforts (**Figure 3**). We know from frequent field visits that planting efforts in the coastal zone of Demak have been initiated by various organizations on a yearly or two-yearly basis. In most cases, this involved large-scale planting on exposed, partly low-lying coastal mudflats (e.g. **Figure 9**, white polygon), or at sites that were already colonized by pioneers (e.g. **Figure 9** 2013, 2015, behind breakwater). The use of natural recruits as an indicator for a sites' suitability for mangrove planting in the coastal zone is a common practice in community-based restoration efforts (Wodehouse and Rayment, 2019), but has also been reported to be unnecessary 70% of the time, as mangroves are already colonizing those sites naturally (Wodehouse and Rayment, 2019). In the case of Demak's breakwater, the practice has led to a mangrove stand of extremely dense and stunted *R. mucronata* trees, with a few surviving *A. alba* trees that rise head and shoulders above the rest (**Figure 9**, top, 2017 and 2018). This illustrates that planting (especially of non-pioneer species) at already newly-colonized sites, is a poor practice.

Most of the large-scale plantations that were done at tidal levels below the threshold of natural recruitment in Demak disappeared before our field visit in a following year. In the literature, there have been examples of successful planting on unvegetated foreshores in the coastal zone (Saenger and Siddiqi, 1993; Uddin and Hossain, 2013). For example, a series of large-scale afforestation projects on newly accreted foreshores in Bangladesh successfully planted an area of 120,000 ha with several mangrove species in the 1980s, including pioneer species such as *Sonneratia apetela* and *Avicennia officinalis* (Saenger and Siddiqi, 1993). The elevated foreshores on which the mangroves were planted were subject to complete desiccation during dry seasons and would normally have been colonized by salt tolerant grasses. One reason for the reported high survival rates of 52% might therefore be that the supply of propagules from pioneer species was the limiting factor for natural mangrove expansion in these projects. These examples suggest that planting, as the sole restoration method, in the coastal zone is only useful if it is done to reintroduce species that are moreover suitable to colonize the site (**Figure 8**). In Demak, natural mangrove cover showed a linear increase on the satellite images, indicating that propagule limitation of pioneer species was not an issue. However, the cover of natural mangroves seemed to stabilize in recent years (**Figure 3**), suggesting that most of the suitable mangrove habitat has now been occupied.

4.2.2 Q3. EMR-Dams: Can EMR-Dams Induce Natural Mangrove Recovery at Coastal Sites?

The aimed-for effect of the EMR-dams in our study area was to expand the existing mangrove habitat by means of wave attenuation, so that the resulting sediment deposition would raise the bed level above the threshold for natural colonization. This method has been effective along several eroding mangrove mud-coasts around the globe (Winterwerp et al., 2020; **Figure 8**). However, along the subsiding coastline of Demak, EMR with the use of permeable structures did not result in large-scale mangrove establishment over the years that the plots were monitored, even though a few seedlings were able to colonize the EMR-sites. It is possible that seedling establishment did occur at a larger scale at the EMR sites, but that the seedlings had disappeared by the time we assessed the plots. This would be in accordance with the findings of Cado et al. (2021), who found that there was abundant seedling establishment behind the dams, but that these seedlings did not persist longer than a few months and did not grow into mature mangroves. This rapid mortality of seedlings could reportedly be explained by a sequence of two processes: 1) the permeable dams do elevate the bed level of the targeted coastal sites, especially during the wet season (Winterwerp et al., 2020), but 2) the whole area is subsequently subject to rapid land subsidence (Kuehn et al., 2009). Hence, the freshly accumulated sediment might initially surpass the elevation threshold for seedling establishment during the wet season, but later sink below the threshold of survival, thus killing the young mangrove seedlings. EMR-dams as a restoration method alone therefore appear not to be enough to support mangrove recovery in this severely subsiding coastal area (**Figure 8**). For mangrove recovery to be successful in Demak, subsidence urgently needs to be addressed, especially by halting ground water extraction (Nicholls et al., 2021).

4.2.3 Q4. EMR-Dams + Planting: Can Mangrove Planting Combined with EMR-Dams Accelerate Recovery?

Despite the low success in natural recruitment at the EMR sites, EMR did increase the survival of established (planted) seedlings (**Figure 6B**), even at sites where natural recruitment behind the dams was not yet taking place (**Figure 7A**). This suggests that a combination of planting and EMR might be useful to overcome (temporary) thresholds of establishment and accelerate revegetation of challenging sites in the coastal zone. However, species choice appears an important factor in the development of the revegetated site after initial recruitment. *R. mucronata*, the species that is most often used in large-scale planting (Wodehouse and Rayment, 2019), proved to have the highest initial survival rate (**Figure 7A**), which might explain the popularity of these species in many voluntary planting projects (Primavera and Esteban, 2008; Wodehouse and Rayment, 2019). However, the slow growth rate of *R. mucronata,* such as observed at this muddy coast (**Figure 7B**), might lead to a mangrove stand that is, more stunted and/or less regenerative than when a pioneer species appropriate to this site would have been used. This outcome has indeed been observed in various studies (e.g. Barnuevo et al., 2017; Fickert, 2020; Proisy et al., 2018), as well as behind the breakwater in Demak (**Figure 9**). It should be noted that in carbonate systems, *Rhizophora* species have been observed as pioneers of the coastal zone [e.g. (McKee, 1993; Piou et al., 2006; Prabakaran et al., 2021)]. This illustrates that it is important to look at natural example sites with a similar biophysical typology as the restoration site [e.g. minerogenic versus organogenic soil (Worthington et al., 2020)], when selecting an appropriate pioneer species. The presence of a few fruiting pioneers can reportedly revegetate a site more efficiently than a planting effort with better surviving, but slower growing species (Fickert, 2020). Although we were not able to include formerly native pioneers such as *Sonneratia alba* and *Aegialitis annulata* (Balun, 2011; Ilman et al., 2016), our Q4 experiment indeed demonstrated that planting with pioneers can quickly lead to the presence of a few reproductive young trees. This illustrates that the trade-off between a species' survival and growth rate should be taken into consideration when choosing species to plant in combination with EMR-measures, preferably using pioneer species at newly accreted sites (**Figure 8**).

5 CONCLUSIONS AND MANAGEMENT IMPLICATIONS

Although the coastline of Demak is a rather extreme case of erosion and land-subsidence, this study does give some useful insights regarding when and how planting and EMR are likely (un)successful. Through a combination of GIS, propagule monitoring, and planting experiments, our results show that:

1) Even in areas with mainly active aquaculture, there is potential for ecological mangrove restoration in ponds lining creeks and rivers, as long as there is a propagule source upstream/upwind. EMR measures in active ponds along rivers could include sluice gate management during the most optimal season (in terms of wind and propagule availability), in combination with partitioning of the pond by adding an inner dike to the river-side of the pond and allowing that part to fill up with sediment so that pioneers can colonize (i. e. greenbelt ponds). Planting may be done on pond bunds or at locations where propagule supply of the target species is limited.

2) Seaward mangrove expansion through planting without additional EMR measures to restore mangrove habitat is likely to be unsuccessful (at low-lying sites) or unnecessary (at proper elevations with existing seedlings), unless species need to be (re) introduced. EMR measures to restore mangrove habitat through sediment trapping along eroding mud-coasts could include placement of permeable (brushwood) dams. Along subsiding coasts, these dams may however not be enough to maintain the bed level above the threshold for mangrove establishment. However, mangrove planting with fast growing pioneer species in combination with sediment trapping by EMR-dams could accelerate mangrove recovery at such sites.

DATA AVAILABILITY STATEMENT

Data in support of this article are available at: https://doi.org/10.4121/17927198.

AUTHOR CONTRIBUTIONS

CB: writing-original draft, experimental design and execution, data analysis, student supervision. AD: experimental design and execution, funding acquisition, field data collection, review, conceptualization, supervision. TB: experimental design, funding acquisition, review, supervision. MM: Resources, Investigation (field data collection), writing-review. RP: Resources, Investigation (field data collection), writing-review. JS: Investigation (GIS data collection), Data Curation, writing-review. FT: Resources, Validation, conceptualization, writing-review. BW: Conceptualization, experimental design, Writing-review, Supervision.

FUNDING

This work is part of the BioManCo project with project number 14753, which is (partly) financed by NWO Domain Applied and Engineering Sciences, and Engineering Sciences, and co-financed by Boskalis Dredging and Marine experts, Van Oord Dredging and Marine Contractors bv, Deltares, Witteveen + Bos and Wetlands International. The Netherlands Sustainable Water Fund provided partial funding for this work through project number FDW14R14. Co-funding to cover the participation by WMR staff was provided by the Netherlands Ministry of Agriculture, Nature and Food Quality through grant KB 36 #4318300114 and through WMR project # 4311500013.

ACKNOWLEDGMENTS

This work is part of the BioManCo project with project number 14753, which is (partly) financed by NWO Domain Applied and Engineering Sciences, and Engineering Sciences, and co-financed by Boskalis Dredging and Marine experts, Van Oord Dredging and Marine Contractors bv, Deltares, Witteveen + Bos and Wetlands International. The BioManCO project is a collaboration between TU Delft, NIOZ and UNDIP and makes use of the framework set up by Building with Nature Indonesia, a program by Ecoshape, Wetlands International, the Indonesian Ministry of Marine Affairs and Fisheries (MMAF), the Indonesian Ministry of Public Works and Housing (PU) and other partners. Co-funding to cover the participation by WMR staff was provided by the Netherlands Ministry of Agriculture, Nature and Food Quality through grant KB 36 #4318300114 and through WMR project # 4311500013. We thank the WMR internship students Ben ten Oever, Inge Kok, Olav Dobber, Jan Burger, Calvin Damen and Joop Bonnet as well as the UNDIP students Hatta Adi Failusuf, Vita Fitriana Mayasari, Wiwid Andriyani Lestariningsih, Marlia Ulfa and Eldita Amalia for contributing to propagule monitoring and the various planting experiments. WMR further generously provided internship and lodging support for these students. Pak Slamet and Ibu Paini are thanked for providing a place to stay and for their all-around hospitality during the fieldwork periods and Pak Muis' skills as a translator are greatly appreciated. Pak Yus Rusila Noor of Wetlands International, Bogor, is thanked for providing many pieces of missing local information. The local Wetlands field team, led by Pak Eko Has is thanked for logistical support and advice, and Apri Susanto Astra, the Wetlands Coastal Safety Manager is thanked for setting up initial contacts and arrangements for all this work. Jenny Cremer, Jan-Tjalling van der Wal of WMR contributed importantly to the GIS mangrove mapping work.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2021.690011/full#supplementary-material

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frontiers in Environmental Science

ORIGINAL RESEARCH
published: 03 March 2022
doi: 10.3389/fenvs.2022.734993

Resident Perceptions and Parcel-Level Performance Outcomes of Mangroves, Beaches, and Hardened Shorelines After Hurricane Irma in the Lower Florida Keys

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OPEN ACCESS

Edited by:
Borja Gonzalez Reguero, University of California, Santa Cruz, United States

Reviewed by:
Pedro Laterra, Consejo Nacional de Investigaciones Científicas y Técnicas, Argentina
Gary Zarillo, Florida Institute of Technology, United States

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Specialty section:
This article was submitted to Environmental Economics and Management, a section of the journal Frontiers in Environmental Science

Received: *01 July 2021*
Accepted: *25 January 2022*
Published: *03 March 2022*

Citation:
O'Donnell KL, Tomiczek T and Scyphers SB (2022) Resident Perceptions and Parcel-Level Performance Outcomes of Mangroves, Beaches, and Hardened Shorelines After Hurricane Irma in the Lower Florida Keys. Front. Environ. Sci. 10:734993. doi: 10.3389/fenvs.2022.734993

Hurricanes have large and lasting effects along coastlines, representing one of the deadliest and costliest natural hazards. Among the rapidly growing literature on the impacts of hurricanes, an increasing topic of interest is the potential role of mangroves, and other coastal habitats, as nature-based strategies (NBS) for coastal defense. In addition to coastal protection, NBS have been shown to provide many ecological, economic, and social co-benefits. However, few studies have assessed coastal resident perceptions or residential-scale performance of NBS, particularly in the wake of major hurricanes. Through a survey of 288 residents of the Lower Florida Keys 1 year after Hurricane Irma, this paper describes hurricane impacts on mangroves, beaches, and hardened shorelines. Specifically, we measured perceptions of shoreline damage and shoreline effectiveness for coastal protection at the community- and parcel-levels. At the parcel scale, we also measured performance outcomes through the cost to repair or replace residential shorelines. At both community- and parcel-levels, beaches were perceived as the most damaged shoreline type, followed by mangroves, and then hardened shorelines as the least damaged. Specifically at the parcel-level, repair actions were not taken by many residents with a hardened shoreline (43.2%) due to their shoreline receiving no damage. However, when repair actions were taken, the average cost to repair or replace parcel-level mangroves ($64.33 (USD) ± SE 58.08 per meter) was less than hardened shorelines ($105.14 (USD) ± SE 38.57 per meter). Additionally, 44% of residents reported that no repair or recovery actions were needed after the storm for damaged mangroves, whereas when hardened structures were damaged, many required at least minor repairs (29.5%). Mangroves were also perceived as the most effective shoreline for storm protection (54% *very* to *extremely effective*) at the community-level. Our findings indicate local community-level awareness of the storm protective properties mangroves provide but also display a disconnect between perceptions and performance outcomes at parcel scales. Although mangroves cost less to repair and are perceived as the most effective at storm protection, the majority of Florida Keys residents own hardened shorelines. Considering the diverse co-benefits mangroves provide and the local support mangroves have, their conservation and restoration could be well-supported for coastal adaptation.

Keywords: nature-based coastal protection, hurricane impacts, ecological knowledge, socio-ecological systems, living shorelines

INTRODUCTION

With sea level rising, ocean warming, and coastal populations increasing, natural hazards increasingly impact both the biophysical landscape, and human communities situated near shorelines (O'Keefe et al., 1976; Emanuel 2005; Donnelly and Woodruff, 2007; Pielke et al., 2008). Historically, many landowners and other important actors have armored coastlines with hardened infrastructure to counteract coastal hazards, such as erosion, flooding, and storms (National Research Council, 2007; Gittman et al., 2015; Scyphers et al., 2015). Hardened infrastructure, such as seawalls and riprap revetments, protect against erosion but degrade the natural environment (Bilkovic et al., 2016; Gittman et al., 2016). Recently, there has been a rapidly growing interest in nature-based strategies (NBS) for coastal protection since these strategies provide co-benefits for ecosystems and human communities (Scyphers et al., 2011; Spalding et al., 2014; Arkema et al., 2017; Gittman and Scyphers 2017).

Previous studies have found that implementing NBS can buffer coastal areas during storms due to their wave attenuating properties, increased flood storage capacity, and ability to retain sediment (Quartel et al., 2007; Mcivor et al., 2012; Hashim and Catherine 2013; Guannel et al., 2016; Munoz et al., 2018). For instance, previous studies covering the impacts caused by Hurricane Irma show that in residential areas of high inundation, homes with mangrove shorelines experienced less damage than homes with bulkheads, and beaches (Tomiczek et al., 2020). Mangroves provide coastal communities with storm protective properties due to their complex and dense network of roots (Davis and Fitzgerald, 2008). Additionally, NBS provides many ecosystem services, including carbon storage, aesthetics, and juvenile fish habitats (Lane et al., 2005). Recent studies have also highlighted the important connection between a person's surrounding environment and their psychological well-being (Collins et al., 2020). Using Hurricane Irma as a case study, this paper describes residents' perceptions and parcel-scale performance outcomes of shorelines after a major hurricane.

Hurricane Irma made landfall in Cudjoe Key on the morning of September 10, 2017, as a Category 4 storm on the Saffir-Simpson Hurricane Wind Scale (National Oceanic and Atmospheric Administration (NOAA) 2017). Hurricane Irma was a large storm with maximum sustained wind fields of 58 m/s (130 mph) extending up to 128 km (80 miles) from the eye at its time of landfall in the Florida Keys. The inundation seen throughout the Lower Florida Keys ranged from 1.5 to 2.4 m (5–8 ft) above ground. Hurricane Irma was ranked as the 5th costliest storm to make landfall in the United States as of 2021, with economic damages estimated at $52.5 billion (USD) (NOAA National Centers for Environmental Information, 2021). The Florida Keys were closed to tourism for approximately 3 weeks, and some areas experienced localized power outages for months. Clean-up of debris from Hurricane Irma was incomplete for nearly 1.5 years after landfall (NOAA 2018). All these day-to-day impacts disrupted residents both economically and psychologically (Lane et al., 2005; Lane et al., 2013; Mamirkulova et al., 2020; Abbas et al., 2021).

Promoting the resilience of coastal communities exposed to intense hazards like hurricanes requires an understanding of the interactions among local biophysical, social, and economic landscapes (Collins et al., 2011). Therefore, coastal planners and scientists must consider the perceptions and decisions made by local residents to have resilient and sustainable systems. This paper describes coastal homeowner perceptions of post-storm impacts to better understand damages to shorelines and the role different types of shorelines play in coastal storm protection. This case study surveys residents along mangroves, beaches, and hardened shorelines in the Lower Florida Keys, who are facing sea level rise, and intense hurricanes along with a growing population and a large tourism industry.

MATERIALS AND METHODS

Our study assessed the impacts of a major hurricane in the Lower Florida Keys through the lens of 288 residents. A parcel-scale survey was used to better understand how residents in the Lower Florida Keys valued their coastal environments (mangroves, beaches, and hardened structures) in the wake of a hurricane.

The people of the Lower Florida Keys are on the forefront of global climate change and hazards. Hurricanes are a normal occurrence in the Lower Florida Keys and many residents know and understand hurricanes because of this (Radabaugh et al., 2019). Since 1852, 60 tropical storms have passed through a 50-mile radius of the Lower Florida Keys, 12 of which have been a major hurricane (category 3 or higher) (Historical Hurricane Tracks, 2021). Previous studies designated the Florida Keys with elevated risks of sea level rise, flooding, and hurricanes (Emrich and Cutter 2011). The Lower Florida Keys are about 177.5 km^2, with 1,531.6 km of shorelines comprised of 63% mangroves, 1% beaches, and 12% hardened structures. However, when limited to the residential areas of the Lower Florida Keys, 78% of residential parcels are along hardened structures, 2% are along beaches, and 15% are along mangroves. Although there are many mangrove forests and mangrove islands throughout the Lower Florida Keys, most residential properties have hardened shorelines. There are concentrated areas of residential properties from Big Pine Key to Big Coppitt Key, most of which are along canals and along the

TABLE 1 | Survey questions used for measuring impacts of Hurricane Irma. The corresponding figure or table showing the results of the survey questions are in bold.

Spatial scale	Concept	Question	Responses
Community	Shoreline Damage (**Figure 2A**)	Thinking about shorelines near where you live, how much were these types of shorelines damaged during Hurricane Irma? (Mangrove, Beach, Bulkhead, Riprap)	No Damage (1) to Ruined (5)
Community	Shoreline Effectiveness (**Figure 2B**)	How effective are the following types of shorelines at protecting coastal properties from storms?	Not Effective At All (1) to Extremely Effective (5)
Community	Shoreline Damage Cause (Text)	What caused the most damage to nearby shoreline structures?	Wind, Storm Surge, Both, Other
Parcel	Home Damage (**Figure 3**)	How would you describe the impact of Hurricane Irma to the following parts of your home?—Overall	No Damage (1) to Ruined (5)
Parcel	Shoreline Typology (Text)	What best describes the shoreline on your property before Hurricane Irma? *Select all that apply*	Mangrove, Beach, Bulkhead/Seawall, Riprap, Other
Parcel	Shoreline Damage (**Figure 4**)	How would you describe the impact of Hurricane Irma to your shoreline?	No Damage (1) to Ruined (5)
Parcel	Shoreline Damage Cause (**Table2**)	What caused the most damage to your shoreline?	Wind, Storm Surge, Both, Other
Parcel	Shoreline Recovery Cost (**Table3**)	What was the cost of repairing or replacing your shoreline?	Dollars, Hours of Your Time
Parcel	Shoreline Maintenance Cost (**Table3**)	On average, how much would you say you spend per year to maintain your shoreline before Hurricane Irma?	Dollars, Hours of Your Time
Parcel	Shoreline Recovery Plans (**Table4**)	What are you doing with your shoreline after the impact of Hurricane Irma?	Nothing shoreline was not impacted, Nothing but shoreline was impacted, Minor repairs, Major repairs or rebuilding to as before, Rebuilding as a different structure
Parcel	Shoreline Rebuilding Change (**Table4**)	What type of shoreline are you rebuilding to?	Mangrove, Beach, Bulkhead/Seawall, Riprap, Other

coast. Out of the 8,614 non-vacant residential parcels 74.3% were waterfront.

Survey Design

A 76-question survey instrument was developed using knowledge from previous engineering, sociological, and environmental studies of disasters (Adger et al., 2016; Scyphers et al., 2019; Tomiczek et al., 2020). Drawing from recent engineering studies of recording hurricane damage to shorelines, a key series of questions in our survey measured shoreline damage in two parts (Tomiczek et al., 2020). First, residents were asked about shorelines near where they live (community-level shorelines), then waterfront residents were asked about the shorelines they own (parcel-level shorelines) (**Table 1**). The types of community-level shorelines were bulkheads (or seawalls), riprap (rocks or coral rock) revetments, beaches, and mangroves. Parcel-level shoreline types were classified into mangrove, hardened, and hybrid (**Supplementary Table S1**). Homes with a hardened shoreline type were associated with residents who reported owning a bulkhead/seawall or riprap. Mangrove shoreline types were shorelines with only mangrove forests (of any size). Hybrid shoreline types were shorelines with both hardened structures and mangroves. Beaches were uncommon along waterfront residences. The survey included a picture of each shoreline type as a reference aid. The survey provided space for residents to describe and report on both community- and parcel-level shorelines not listed within the survey. Respondents reported all shoreline damage using a 5-point Likert scale (*No Damage, Lightly Damaged, Moderately Damaged, Majorly Damaged,* and *Ruined*) (Tomiczek et al., 2020). Respondents also reported the effectiveness of each shoreline type at protecting coastal properties from storms using a 5-point Likert scale (*Extremely Effective, Very Effective, Moderately Effective, Slightly Effective,* and *Not Effective at All*). Waterfront respondents then reported how much their parcel-level shoreline cost to maintain per year and to repair or replace their shoreline after the storm. Finally, residents reported on the primary cause of damages to community- and parcel-level shorelines (*Storm surge, Winds,* and *both*). We excluded blank responses for monetary questions from calculations and tests. Many residents were still waiting to receive compensation for their insurance claims, had not yet finished repairing their property, or did not want to report on monetary outcomes.

The survey instrument also measured overall home damage using a 5-point Likert scale (*No Damage, Lightly Damaged, Moderately Damaged, Majorly Damaged,* and *Ruined*) which is used to test the ability of shorelines to buffer storm impacts. Housing characteristics were collected based on engineering assessments (*elevated, number of stories,* and *building material*) and vulnerability assessments (*housing ownership, housing type,* and *primary residence*) (Tomiczek et al., 2014; Tomiczek et al., 2020). Finally, the survey included questions to document gender, age, annual household income, education, and years lived at current residence.

Survey Data Collection

Approximately 1 year after the storm (June 2018–November 2018) a mixed-mode (online, physical) parcel-scale survey was sent to residents asking about Hurricane Irma impacts in the Lower Florida Keys (**Figure 1**). A stratified sample of 1,500 residents (1,000 waterfront and 500 non-waterfront) were randomly selected using the publicly available Monroe County tax database

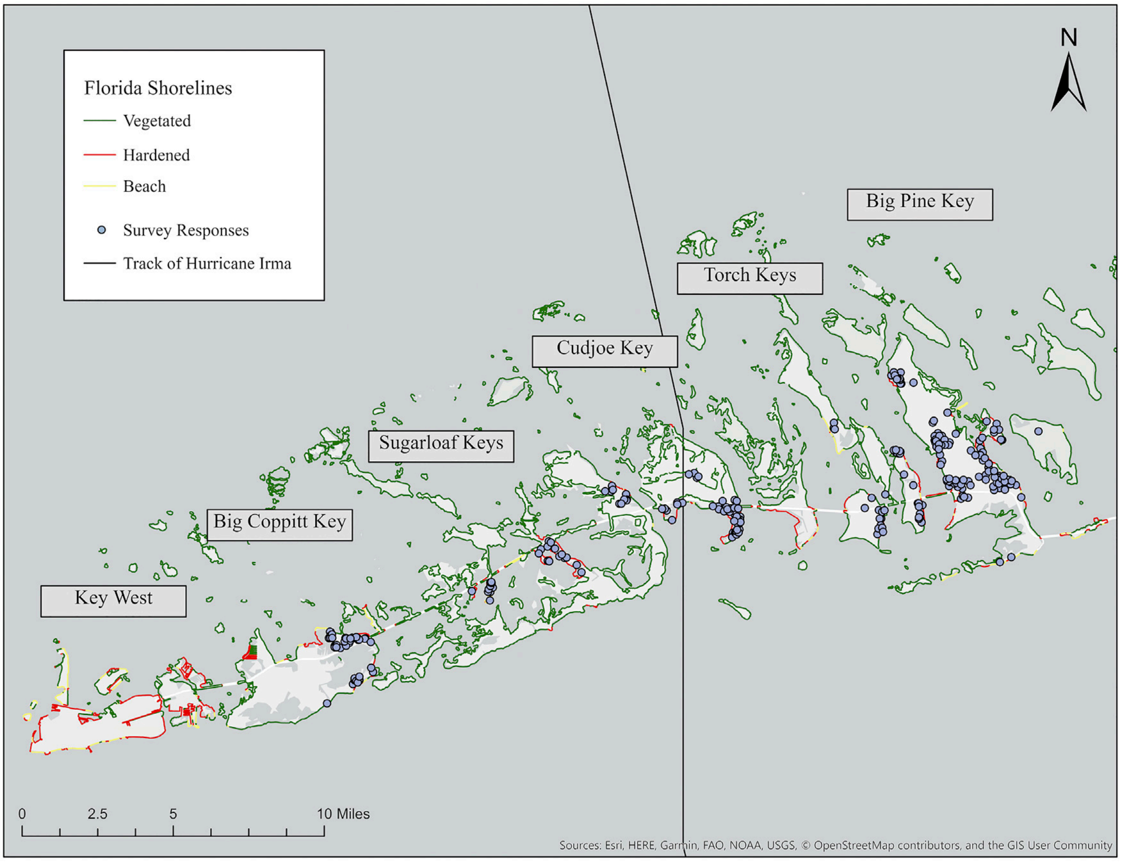

FIGURE 1 | Map of Hurricane Irma's path, shoreline condition, and location of survey respondents throughout the Lower Florida Keys. The map shows the Lower Florida Keys from Key West (not surveyed) on the westernmost side to Big Pine Key on the eastern side of the map. The black line running through Cudjoe Key depicts the track of Hurricane Irma. The NOAA ESI lines reflect the shoreline types based on the following groupings: hardened (seawalls or riprap revetments), vegetated (mangrove, scrub-shrub wetlands, marsh grass, etc.), and beaches. The blue dots represent the survey respondents' locations.

(University of Florida's GeoPlan Center, 2006) and the NOAA Environmental Sensitivity Index (ESI). Similar to the methods described by Dillman et al. (2014), residents received three mailings which provided recipients with both an online and a physical survey option. The survey research firm Qualtrics (Qualtrics, Provo, UT) hosted the online survey, and served as the database for the physical survey. The survey yielded an adjusted response rate of 24%, totaling 288 responses (displayed in **Figure 1**).

Analysis

All responses were analyzed in the Statistical Package for Social Science (SPSS) Version 26. Responses were connected to their surrounding environmental characteristics using ArcGIS Pro 2.4.0. Shoreline data from NOAA's ESI, storm conditions from the Coastal Emergency Risk Assessment (CERA)'s post-storm models, and the island location of residents were linked to each response using unique ID codes. The accuracy of reported location, shoreline type, and length of shoreline were verified using the Monroe County tax database and NOAA's ESI. Using CERA's post-storm models, the storm conditions of maximum winds, and maximum inundation were found for each parcel.

We used multivariate, univariate, and descriptive statistics to evaluate the potential cause of shoreline damage and the relationships of shoreline damage to residents' perceptions and actions (Scyphers et al., 2019). To allow for comparison of shorelines between residents of different parcel sizes, the cost of shoreline maintenance, and the cost of the damage from Hurricane Irma was divided by the length of shoreline owned. Using a Kruskal–Wallis test, we compared shoreline damage and coastal protection effectiveness across the different Keys. Additionally, we compared perceptions of community-level shorelines by the type of shoreline owned. Next, we compared the parcel-level shoreline damage (monetary values per meter of shoreline and reported damage states) by shoreline type. Finally, we compared parcel-level storm characteristics of maximum inundation and maximum winds from Hurricane Irma by the shoreline damage state. When a Kruskal–Wallis test produced a significant result, a Dunn-test was completed to find the categories producing the significant result. We used a Spearman's rank two-tailed correlation to relate the monetary values of damage with the reported damage states. We also related shoreline damage and coastal protection effectiveness for each shoreline type. All

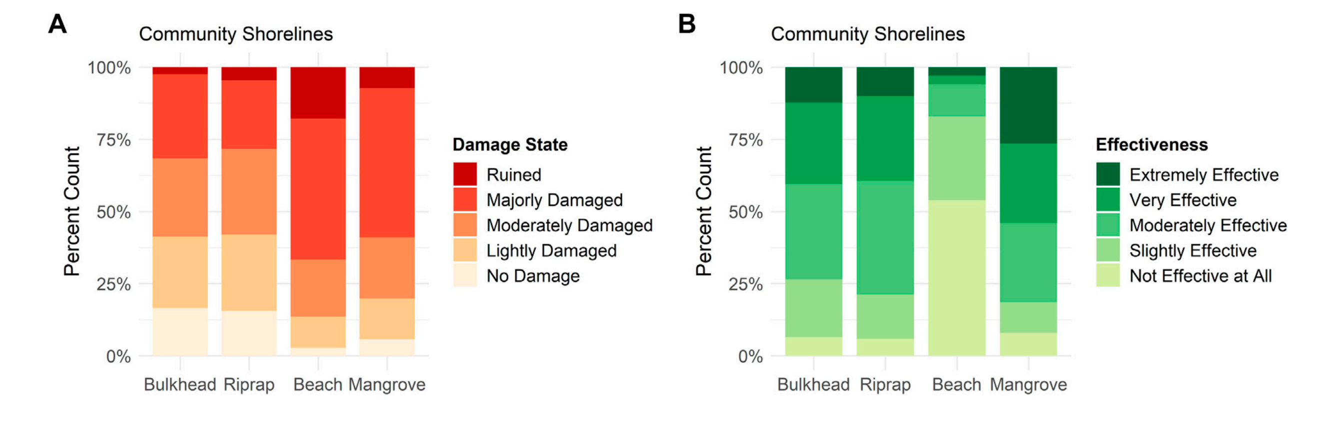

FIGURE 2 | Perceptions of community-level shorelines, both the reported damage state **(A)** and the coastal protection effectiveness **(B)**.

statistical results used an alpha level of 0.05 to indicate significance.

RESULTS

Home Characteristics and Demographics

The proportion of survey respondents from each of the Lower Florida Keys was 14.3% ($n = 41$) in Big Coppitt Key, 11.9% ($n = 32$) in Saddlebunch and Sugarloaf Key, 24.1% ($n = 69$) in Cudjoe Key, 10.8% ($n = 31$) in the Torch Keys, and 38.8% ($n = 111$) in Big Pine Key. Of these coastal residents, 70.8% ($n = 204$) lived in or owned waterfront property (including canals) which aligns with the Lower Florida Keys population. Of these waterfront properties, 74.3% ($n = 150$) had hardened shorelines, 13.4% ($n = 27$) owned mangrove shorelines, and 12.4% ($n = 25$) owned hybrid shorelines.

Participants were primarily white (94%, $n = 251$) and approximately half were female (51.1%, $n = 141$). The average age of respondents was 62.8 ± 13 (SE) years old. Residents primarily reported having between some college (29%, $n = 82$) or a bachelor's degree (35.3%, $n = 100$). The majority of residents (78.4%, $n = 171$) had household incomes greater than or equal to the median household income ($65,747 (USD)) given by the 2017 US census for the Key West FL, micro-area.

Perceptions of Community-Level Shoreline Impacts and Effectiveness

On average, community-level shorelines around the Lower Florida Keys (bulkhead/seawall, riprap, beach, and mangrove) were ranked as *moderately damaged* (Likert scale: 3.17 ± 0.033 SE). Most residents reported storm surge ($n = 149$, 54.4%) as the primary cause of community-level shoreline damage. Damage states were significantly different based on the type of shoreline (Kruskal–Wallis test: H = 131.83, $df = 3$, $p < 0.001$; **Figure 2A**.). The damage states of bulkhead shorelines were not significantly different from riprap shorelines (Dunn Test: $Z = 0.155$, $p = 0.877$). In addition, the two hardened shoreline types were less damaged than mangroves (bulkheads: $Z = -6.646$, $p < 0.001$, riprap: $Z = 6.598$, $p < 0.001$; **Figure 2A**). Community-level beach shorelines were the most damaged, with 49.2% of respondents reporting *major damage* and 18.0% reporting *ruined* (**Figure 2A**). Community-level shoreline damage significantly differed by Key for all shoreline types except bulkheads (Kruskal–Wallis tests; Mangrove: H = 30.677, $df = 4$, and $p < 0.001$; Beach: H = 36.216, $df = 4$, and $p < 0.001$; Bulkhead H = 6.020, $df = 4$, $p = 0.198$; and Riprap H = 18.761, $df = 4$, and $p < 0.001$). Mangroves, beaches, and riprap all had higher damages reported by those who resided in Big Pine Key.

Shoreline coastal protection effectiveness was also significantly different by shoreline type (Kruskal–Wallis test: H = 321.61, $df = 3$, and $p < 0.001$; **Figure 2B**.). Mangroves were ranked the most effective shoreline type at protecting coastal properties from storms, with 54.0% of respondents reporting the protection provided was *very* to *extremely effective*. Conversely, beaches were the least effective, with 50% reporting *not effective at all*. The effectiveness of bulkheads and riprap were not significantly different (Dunn Test: $Z = 0.262$, $p = 0.793$). Shoreline coastal protection effectiveness did not vary by key for all shoreline types except Riprap (Kruskal–Wallis tests; Mangrove: H = 7.220, $df = 4$, and $p = 0.125$; Beach: H = 5.923, $df = 4$, and $p = 0.205$; Bulkhead H = 2.709, $df = 4$, and $p = 0.608$; and Riprap H = 12.157, $df = 4$, and $p = 0.016$). Respondents in Big Coppitt Key perceived riprap as slightly less effective. For waterfront homeowners, perceptions of community-level shoreline effectiveness were not significantly different for the type of shoreline owned (Kruskal–Wallis tests; Mangrove: H = 2.761, $df = 3$, and $p = 0.430$; Beach: H = 3.789, $df = 3$, and $p = 0.285$; Bulkhead H = 2.598, $df = 3$, and $p = 0.458$; and Riprap H = 0.482, $df = 3$, and $p = 0.923$).

Perceptions and Performance of Parcel-Level Shorelines

Most residents reported their home as *moderately* or *lightly* damaged, 39.2 and 30.9% respectively. On average, households spent $35,611.06 (USD) (± 4,487.73 SE) to repair or replace their overall home damage. As expected, the cost reported to repair or replace aspects of a home was significantly different based on the reported damage state (Kruskal–Wallis: H = 41.935, $df = 4$, and $p < 0.001$). Home damage states were similar across the keys (Kruskal–Wallis: H = 9.301, $df = 4$, and $p = 0.054$; **Figure 3A**). However, Big Pine Key had

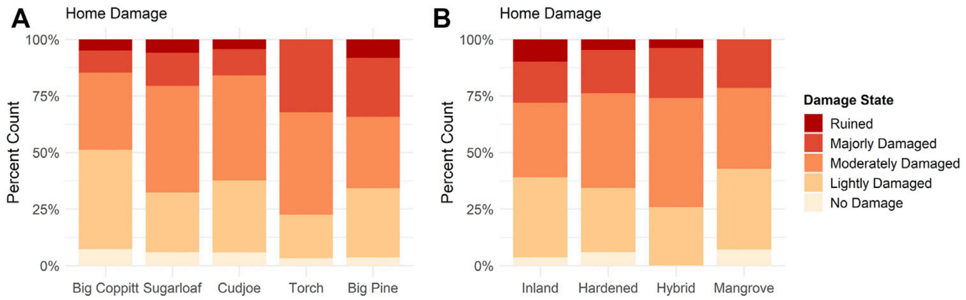

FIGURE 3 | Perceptions of home damage states. The colors within each bar reflect the proportion of each reported home damage state. **(A)** shows the percentage of each reported damage state by key and **(B)** does the same by shoreline type.

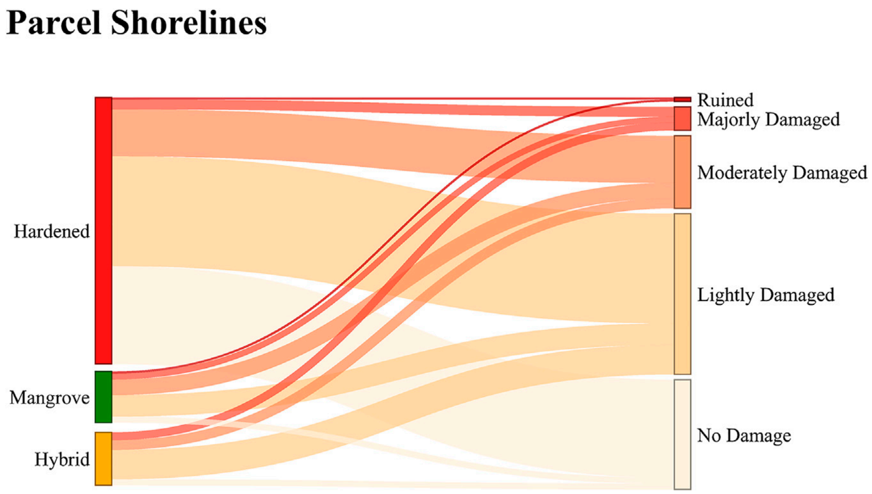

FIGURE 4 | Sankey diagram of parcel-level shoreline damage states. Number of responses is proportional to the height of the bars.

TABLE 2 | Percentage of respondents (and number in parentheses) reporting the primary damage type caused by Hurricane Irma displayed across parcel-level shoreline types (Chi-squared test: χ^2 = 6.323, df = 4, p = 0.176).

	Shoreline types			
Damage Type	**Hardened**	**Hybrid**	**Mangrove**	**Total**
Wind	20.7% (17)	12.5% (3)	37.5% (9)	22.1% (29)
Storm Surge	69.5% (57)	75.0% (18)	45.8% (11)	66.4% (87)
Other	9.8% (8)	12.5% (3)	16.7% (4)	11.5% (15)
Total	82	24	24	131

the most reports of *ruined* or *majorly* damaged homes, 8.1% and 26.1% respectively. Overall home damage of waterfront homeowners was not significantly different from inland residents (Kruskal–Wallis: H = 0.037, *df* = 1, and p = 0.848; **Figure 3B**). For the subset of only waterfront homes, home damage states were not significantly different across shoreline type (Kruskal–Wallis: H = 1.506, *df* = 2, and p = 0.471; **Figure 3B**).

Overall, parcel-level shorelines fared well during the storm, with the plurality of respondents reporting *lightly damaged* shoreline states (43.1%, n = 82; **Figure 4**). On average, shoreline repair costs were $4,884.29 (USD) (± SE 1,121.54). Residents reported storm surge as the primary cause of parcel-level shoreline damage (66.4%, n = 87; **Table 2**). Residents who specified another cause of damage often reported that wind (or tornadoes) and storm surge equally caused damage (n = 14). Homeowners of hardened and hybrid shoreline types reported that storm surge caused the most damage (hardened 69.5%, n = 57; hybrid 75.0%, n = 18; **Table 2**). However, only about half of mangrove shoreline homeowners reported storm surge as the primary cause of damage (45.8%, n = 11; **Table 2**). The cause of damage was not significantly different by shoreline type (Chi-squared test: χ^2 = 6.323, df = 4, and p = 0.176).

Although the primary cause of damage reported was storm surge, damage states of residential shorelines were similar across storm characteristics. Residents across the Lower Florida Keys experienced winds between 49.2–53.6 m/s (110–120 mph). Although winds were similar across parcel-level damage states, ruined shoreline types experienced slightly higher winds (Kruskal–Wallis: H = 8.717, *df* = 4, and p = 0.069). Residents of the Lower Florida Keys experienced an average inundation of 1.30 m (4.27 ft) and a range of 0.18–2.47 m (0.59–8.10 ft).

TABLE 3 | Reported costs of maintaining and repairing parcel-level shorelines per meter. The first row in the table displays the average parcel-level shoreline length in meters for each category of shoreline type. Next row is the average (± SE) per meter annual maintenance cost of parcel-level shoreline for each category of shoreline type reported in US dollars. Next row is the average (± SE) per meter cost to repair or replace parcel-level shorelines for each category of shoreline type. Finally, the last row is the combination of these two costs per meter for each shoreline type.

	Shoreline type			
	Hardened	Hybrid	Mangrove	Total
Shoreline Length (m)	24.29	41.59	50.63	29.61
Yearly Cost to Maintain per meter of shoreline (USD) (Mean ± SE)	\$21.39 ± 6.00	\$11.90 ± 3.74	\$11.60 ± 6.76	\$18.63 ± 4.53
Cost to repair or replace per meter of shoreline (USD) (Mean ± SE)	\$105.14 ± 38.57	\$66.89 ± 30.90	\$64.33 ± 58.08	\$92.54 ± 28.14
Repair + Maintain cost (2017 expense) per meter of shoreline (USD) (Mean ± SE)	\$84.56 ± 24.72	\$66.81 ± 28.15	\$52.54 ± 37.69	\$76.81 ± 18.97

TABLE 4 | Repair and recovery actions residents took or will take in response to parcel-level shoreline damage. The table displays the percent of reported actions (and number in parentheses) across the shoreline type owned. The row above the Totals display the planned structures when respondents are rebuilding their shoreline as a different structure.

Repair and recovery actions	Shoreline type			
	Hardened	Hybrid	Mangrove	Total
Nothing, Shoreline was not impacted	43.2% (57)	11.1% (3)	24.0% (6)	36.0% (67)
Nothing, Shoreline was impacted	14.4% (19)	11.1% (3)	44.0% (11)	17.7% (33)
Minor repairs	29.5% (39)	55.6% (15)	24.0% (6)	32.8% (61)
Major repairs or rebuilding as before	12.1% (16)	18.5% (5)	4.0% (1)	11.8% (22)
Rebuilding as a different structure	0.8% (1)	3.7% (1)	4.0% (1)	1.6% (3)
(old shoreline to new shoreline)	Bulkhead to Riprap	Bulkhead/Mangrove to new dock structure	Mangrove to Bulkhead/Riprap	
Total	132	27	25	186

Inundation was similar across parcel-level shoreline damage (Kruskal–Wallis H = 3.666, df = 4, and p = 0.453). Additionally, the various shoreline types experienced similar levels of inundation (Kruskal–Wallis: H = 5.036, df = 2, and p = 0.081) and winds (Kruskal–Wallis: H = 0.970, df = 2, and p = 0.616) across the study area. However, parcel-level shoreline damage was significantly different by shoreline type (Kruskal–Wallis: H = 13.883, df = 2, and p = 0.001; **Figure 4**). Many hardened shorelines had low damage states, with 37.0% reporting *no damage* and 40.7% reporting *lightly damaged*. Respondents mostly reported mangrove and hybrid shorelines as *lightly damaged* (42.3 and 55.6% respectively), and few reported these shorelines as *no damage* (11.5 and 11.1%).

The cost to repair per meter of shoreline was positively correlated with reported shoreline damage states (Spearman's correlation: R = 0.640, p < 0.001). On average, residents with mangrove shorelines had the longest shoreline property (50.63 m.; **Table 3**). The maintenance cost per meter of shoreline was similar for both hybrid (\$11.90 (USD) ± SE 3.74) and mangrove (\$11.60 (USD) ± SE 6.76) shorelines. Hardened shorelines, had the highest per meter cost of maintenance (\$21.39 (USD) ± SE 6.00). Hardened shorelines also had the highest per meter cost for shoreline repair (\$105.14 (USD) ± SE 38.57). When adding together the yearly maintenance cost and the cost to repair or replace the shoreline, mangroves were the least expensive per meter (\$52.54 (USD) ± SE 37.69; **Table. 3**).

Shoreline recovery actions varied by shoreline type (Chi-squared test: χ^2 = 26.566, df = 6, p < 0.001; **Table 4**). Most homeowners with mangroves did not take action to repair their shoreline (n = 17, 68.0%), although most homeowners reported their parcel-level mangrove shoreline as *lightly damaged* (**Figure 4**). Most residents with a hardened shoreline reported that no repair actions were needed due to their shoreline not being impacted (n = 57, 43.2%), aligning with the low damage states of hardened shorelines discussed above (**Figure 4**). If a resident's shoreline was impacted, most hardened and hybrid shoreline owners reported minor repairs (hardened n = 39, 29.5%; hybrid n = 15, 55.6%; **Table 4**). Of the 186 waterfront residents with damage to their shoreline, only 3 reported an intention to rebuild with a different structure.

DISCUSSION

Our study provides insight into the performance and community perceptions of mangroves, beaches, and hardened shorelines in the Lower Florida Keys following Hurricane Irma. A key finding of our study is that storm damage varied across different types of shorelines. Residents reported mangroves had higher damage states than hardened shorelines at both the parcel- and community-levels. However, monetary impacts and reported recovery actions show that mangroves performed well during and after the storm. The per meter cost to repair or replace mangroves was less than hardened shorelines, and fewer repair actions were necessary when damage occurred. Additionally, respondents reported on the effectiveness of mangroves at protecting the coast from storm conditions. Both cost and effectiveness have been ranked highly as attributes homeowners consider when making decisions about their shorelines (Scyphers et al., 2015; Smith et al., 2017). Therefore, our findings could make mangrove systems a valuable shoreline option for coastal homeowners in the Lower Florida Keys. However, the high percent of residents with hardened shorelines, and the strong preference of homeowners to repair or recover their shorelines to its pre-storm condition, indicates that perceptions of mangroves and shoreline actions are not aligning. Therefore, additional incentives may be needed for promoting NBS (Scyphers et al., 2020).

Respondents reported storm surge as the primary cause of damage to parcel- and community-level shorelines. Previous studies on Hurricane Irma, and general post-storm models, show that areas on the eastern side of the storm experienced higher inundation and greater wind speeds (NOAA 2017; Tomiczek et al., 2020), which aligns with the greater damage reported to the east. Interestingly, the bulkhead damage was not different by Key, whereas other shoreline types were most damaged on the eastern-most island of Big Pine Key. The variation in mangrove damage across islands suggest that mangrove shorelines may be more resilient than hardened shorelines at lower inundation values and slower wind speeds (Constance et al., 2021). This is further supported by the high damages to mangroves reported in areas of high inundation and high wind speeds (east of the storm). Conversely, hardened shorelines have similar levels of damage across all Keys, and all storm characteristics.

Although mangroves attenuate waves (Smith et al., 2009; Narayan et al., 2016; Tomiczek et al., 2020), these shorelines allow the water to flow through the roots and may cause more visually apparent damages. Specifically, storm surge can bring sediment and human-made debris into the complex root system of mangroves, sometimes creating delayed mortality of mangrove systems after a storm (Smith et al., 2009; Radabaugh et al., 2019). Hurricane winds can also strip leaves off mangrove trees causing additional visual damage and makes other damages more apparent (Smith et al., 2009). These visible impacts were described throughout several responses and display important influences on residential perceptions of damage states across various shoreline types. However, the monetary costs and reported repair actions of parcel-level mangroves indicate that these visual damages were less costly to remediate.

Previous studies have shown the importance of cost and durability when homeowners make decisions about shoreline protection (Scyphers et al., 2015; Smith et al., 2017). Although mangroves had higher damage states than hardened structures, they also cost less to repair or replace per meter of shoreline (**Table 3**). This finding could be due to a natural shorelines ability to grow back after a disturbance without human intervention (Radabaugh et al., 2019). Most waterfront residents with mangroves reported that although their shoreline was impacted, no further actions were needed to repair or replace their shoreline, further supporting the ability of mangrove stands to grow back. Conversely, hardened shorelines cost more to repair and replace, and most residents with hardened shorelines reported that if they received damages, minor to major repairs were needed.

Mangroves also cost less to maintain (**Table 3**), because mangrove shorelines can mature, strengthen, and even expand over time in the right conditions (Spalding et al., 2014; Constance et al., 2021). Whereas hardened structures are rigid and face degradation over time (Sutton-Grier et al., 2018). Previous studies have also shown higher costs for homeowners with hardened structures compared to vegetated shorelines (Gittman and Scyphers 2017; Smith et al., 2017). The economic benefit that natural shorelines provide must be highlighted since cost is important to coastal homeowners (Scyphers et al., 2015; Gittman et al., 2016; Gittman and Scyphers 2017). With home damage typically being the highest priority for repairs, a shoreline that allows for lower maintenance is highly important to homeowners. However, a longer post-storm period would be needed to fully account for shoreline impacts and recovery as many residents were still working on damaged aspects of their property at the time of the survey and could not yet report on the full recovery costs or actions.

As stated above, mangroves were perceived to be slightly more damaged than hardened shorelines, and mangroves were seen as the most effective shoreline at protecting coastal property. Similarly, Furman et al. (Furman et al., 2021) found that residents in Key West perceived mangroves as more beneficial at mitigating storm impacts. Residents view mangroves as an effective coastal protection shoreline type although most residents own hardened shoreline structures. These perceptions match post-storm engineering measurements where homes with mangroves present were better protected at higher inundation levels (Tomiczek et al., 2020). This survey disagreed with the engineering assessment, finding that home damage states were not significantly different across shoreline types (**Figure 3**). However, many majorly damaged and ruined homes (seen in other Hurricane Irma reports) may not have been reported due to displacement, causing our survey to have slightly skewed damage states (NOAA 2017; Tomiczek et al., 2020). Given the similar home damages reported across shoreline types in this study, mangrove shorelines provided similar levels of coastal protection to hardened and hybrid shorelines during Hurricane Irma.

While local knowledge on the benefits of mangroves is prevalent in the Lower Florida Keys, most homeowners own hardened shorelines. Additionally, very few homeowners reported an intention to rebuild their damaged shoreline as a different structure. Some theories on environmental decision-making rely on cognitive fixes to change human behavior, claiming that providing more information about a topic can cause people to change their decisions, and actions (Heberlein 2012). However, residents ranked mangroves as the most effective shoreline type at providing coastal protection. Therefore, other limiting factors may be present when these homeowners make decisions about their shorelines (Scyphers et al., 2015; Smith et al., 2017). Recent studies show that NBS projects were successful when including additional efforts, strategies, and funding (DeAngelis et al., 2020; Scyphers et al., 2020). DeAngelis et al. (2020) highlights the importance of political motivation and funding along with public understanding and demand for successful restoration projects. At the residential scale, Scyphers et al. (2020) also found an initially low response of residents willing to change their hardened shoreline to NBS during a window of opportunity. However, they found that a modest economic incentive greatly increased the likelihood of rebuilding with NBS. The diverse co-benefits of mangrove shorelines, coupled with their the effectiveness for coastal protection, provides support for including mangroves in coastal protection planning, and implementation.

CONCLUSION

Coastal residents are key decision-makers for conserving and restoring mangroves along shoreline structures. The local knowledge and experiences measured in our study can provide coastal planners, scientists, and other government agencies with important insights for coastal conservation and climate adaptation. Previous studies have highlighted the importance of sharing and using diverse stakeholder knowledge when building more sustainable management strategies for social-ecological systems (Aminpour et al., 2021). This survey provides support for mangroves as a coastal protection strategy, finding that mangroves performed well during the storm, and cost less time and money during recovery efforts. In addition to these findings, other studies display environmental, social, and economic co-benefits of mangroves (i.e., carbon storage, aesthetics, and juvenile fish habitats, etc.). However, in many urban areas vegetated shorelines are not widely maintained and often converted to hardened shorelines. Future studies are needed to understand and overcome barriers to conserving, restoring, and implementing mangroves and other types of NBS along residential shorelines.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusion of this article will be made available by the authors, without undue reservation.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by the Northeastern University's Institutional Review Board. The patients/participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

All authors contributed to the conception and design of the study. KO organized the database, performed the statistical analysis, and wrote the first draft of the manuscript. All authors contributed to manuscript revisions. All authors have read and approved the submitted version.

FUNDING

This study was funded by Northeastern University, the United States Naval Academy, and the National Science Foundation under Grant No. 2110262. All human subjects research was approved by Northeastern University's Institutional Review Board (IRB #120517). Any opinions, findings, and conclusions or recommendations expressed in this material are those of the author(s) and do not necessarily reflect the views of Northeastern University, the United States Naval Academy, or the National Science Foundation.

ACKNOWLEDGMENTS

We are grateful to the coastal residents who participated in our study. We thank S Gibbs and N Card for assistance with survey deployment and data entry.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2022.734993/full#supplementary-material

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Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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frontiers | Frontiers in Environmental Science

REVIEW
published: 29 June 2022
doi: 10.3389/fenvs.2022.829526

Nature-Based Solutions in Coastal and Estuarine Areas of Europe

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Momentum for sustainable and climate resilience solutions for coastal protection are growing globally given the pressing need to prevent further loss of biodiversity and ecosystems while meeting the climate change adaptation and mitigation goals. Nature-Based Solutions (NbS) represent an opportunity to align environmental and resilience goals, at a time of strained budgets in a global context and when short-term needs may run counter to long-term goals. In Europe, NbS fit the mandates of major EU environmental and climate change policies by restoring biodiversity and enhancing climate-resilience and carbon sequestration. Previous studies have compiled scientific evidence about hydro-meteorological hazards for the use of NbS. However, their implementation at scale is still lacking. As the knowledge and experience with NbS for adaptation to natural hazards and climate change increases, it becomes more important to draw lessons learned and insights for replicating and scaling up NbS, especially in coastal areas where their implementation is still limited compared to other environments. This study analyzed NbS case studies across European coastal and estuarine areas to draw key lessons, understand better the current status of implementation, and identify key challenges and gaps. From a total of 59 NbS case studies associated with flooding, erosion and biodiversity loss, results show an increase in NbS implementation since 1990s, but most rapidly between 2005 and 2015. Most of the case studies are hybrid solutions employing wetlands, predominantly located in the United Kingdom (UK) and the Netherlands. Funding of NbS is largely from public sources, and rarely come from a single or a private source. Three-quarters of the case studies reported monitoring activities, but more than half did not disclose quantitative results related to effectiveness against flooding and/or erosion. The need to improve coastal defenses was indicated as the main motivation for NbS implementation over traditional structures, while sustainability was the most mentioned additional reason. Although a variety of co-benefits and lessons learned was identified, clearer descriptions and enhanced details of such information are required. There is a need for tools and strategies to expand knowledge sharing of lessons learned to enable further replication of successful cases in other areas.

Keywords: nature-based solutions, coastal, estuarine, climate adaptation, coastal protection, sustainability, natural infrastructure, Europe

OPEN ACCESS

Edited by:
Xander Wang,
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Reviewed by:
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Specialty section:
This article was submitted to Interdisciplinary Climate Studies, a section of the journal Frontiers in Environmental Science

Received: *05 December 2021*
Accepted: *06 June 2022*
Published: *29 June 2022*

Citation:
Moraes RPL, Reguero BG, Mazarrasa I, Ricker M and Juanes JA (2022) Nature-Based Solutions in Coastal and Estuarine Areas of Europe. Front. Environ. Sci. 10:829526. doi: 10.3389/fenvs.2022.829526

INTRODUCTION

The impacts of climate change in coastal areas concern a significant part of society given that a large fraction of the global population (41%) and world's megacities (60%) are located in the coastal zone (Martínez et al., 2007). Coastal areas combine high population density, concentration of economic activities (Creel 2003), and high exposure to the impacts of waves, extreme sea levels, runoff, land subsidence and other hazards (Lee et al., 2021). Sea Level Rise (SLR) and impacts from extreme weather events will be most felt in most coastal areas in the next decades (Oppenheimer et al., 2019) where 190–630 million people are predicted to be inundated by 2,100 (Kulp and Strauss 2019). Coastal flood risk is likely to increase due to expected strengthening of storm intensity, accelerated SLR and land subsidence (Reguero et al., 2015; Syvitski et al., 2009; Temmerman et al., 2013; Lin et al., 2012).

In Europe, nearly half of the population lives less than 50 km from the sea (Statistical Office of the European Communities, 2011) and many coastal regions are already experiencing the impacts and costs of climate change and coastal hazards (Masselink et al., 2016; Ganguli and Merz 2019; Madsen, Mikkelsen, and Blok 2019). Most European countries are expected to be affected by frequent flooding events and SLR over the upcoming decades (European Environment Agency, 2019). For example, in the United Kingdom (UK), people exposed to a relevant annual likelihood of coastal flooding would increase between 37% and 178% due to SLR (Edwards 2017). Impacts from coastal flooding across continental Europe are also projected to increase significantly with rising sea levels (Vousdoukas et al., 2018), but increased climate hazards will coincide with an expected increase in population living in coastal areas, by one estimate of 355 million people by 2035 (Maul and Duedall 2019).

The traditional coastal protection approach has relied on 'hard' engineering solutions that are unlikely to withstand the increasing pressure from intensified hydrometeorological hazards caused by climate change (Kumar et al., 2020). Moreover, the maintenance costs of such structures could become unfeasible (Morris et al., 2018). Therefore, the need of lower cost, sustainable and resilient solutions is increasing. In this context, Nature-based Solutions (NbS) are emerging globally as a strategy that employ natural features to address hazards while enhancing biodiversity (EC 2021b). NbS may include actions to protect, sustainably manage and restore natural or modified ecosystems that provide critical ecosystem services for human well-being and biodiversity (Cohen-Shacham et al., 2016). NbS leverage the hazard mitigation properties of natural ecosystems. In coastal environments, ecosystems such as dunes, seagrass meadows, saltmarshes and biogenic reefs (e.g., oyster reefs) are able to protect coastal areas from erosion and flooding by dissipating the hydrodynamic energy through their submerged canopies or structural complexity (Gedan et al., 2011; Temmerman et al., 2013; Hanley et al., 2014; Ondiviela et al., 2014). Unlike "hard" engineering structures, coastal vegetated ecosystems and biogenic reefs can self-adapt to sea level rise through different mechanisms. Vegetated ecosystems are able to enhance soil vertical accretion and soil elevation due to the accumulation of large belowground biomass and the trapping of particles from the water column (Duarte et al., 2013; Kirwan and Megonigal 2013; Potouroglou et al., 2017). Oyster reefs grow vertically through attracting oyster larvae that drift through the water and latch onto the existing wall, contributing to its growing (Rodriguez et al., 2014). In addition, coastal habitats provide multiple other ecosystem services relevant to coastal communities, such as fisheries support, biodiversity, water quality improvement, and recreational and cultural benefits (Barbier et al., 2011). In the case of vegetated ecosystems, they are also significant carbon sinks due to their high productivity and their high carbon burial capacity (Mcleod et al., 2011), playing a significant role in climate change mitigation (Nellemann et al., 2009; Serrano et al., 2019).

In Europe, the European Commission (EC) is devoting great efforts in supporting NbS to address climate change and other environmental challenges (EC, 2015). For instance, the European Green Deal, a roadmap to make the EU's economy sustainable, places NbS at the center of climate adaptation and mitigation and highlights their role in ensuring healthy and resilient seas and oceans. Moreover, the implementation of NbS is also supported by different European policies, such as The Green Infrastructure Strategy (EC, 2021a), the EU Strategy on Adaptation to Climate Change (EC, 2013) or the Floods Directive (2007/60/EC). In addition to the policy framework, the EC has invested substantial financial resources in NbS dissemination, which resulted in the creation of several integrative platforms aiming to support the replication, upscaling and dissemination of NbS (Faivre et al., 2017; Kumar et al., 2020).

The application of NbS to mitigate and adapt to climate change in coastal and estuarine areas also provides an opportunity to restore and maintain coastal ecosystems in Europe, which have been historically threatened and transformed by human activities with an estimated reduction in their original surface of 2/3 for coastal wetlands (Airoldi and Beck 2007). The destruction of coastal ecosystems leads to the loss of all ecosystem services provided, including the role these ecosystems play in coastal protection against climate change hazards (Vo et al., 2012). The application of NbS can lead to the recovery and maintenance of biodiversity and all other coastal ecosystems services provided to societies (Faivre et al., 2017), while contributing to meet the goals of other conservation policies (e.g., EU Habitats directive; EU Birds Directive; Esteves 2014).

Despite the policy tailwinds, the application of NbS for coastal protection is still scarce compared to traditional engineered options in most of countries worldwide, including Europe (Morris et al., 2018). Major barriers for the wider implementation of NbS are the difficulty to predict its long-term effectiveness, the lack of standardized methods to assess efficacy, and a lack of data to produce cost-benefit analysis, especially when compared to traditional engineering approaches (Temmerman et al., 2013; Narayan et al., 2016; Morris et al., 2018). NbS in the context of hydrometeorological hazards has been the focus of extensive research during the last decade (e.g., Arkema et al., 2017; Faivre et al., 2017; Debele et al., 2019; Kumar et al., 2020; Kopsieker et al., 2021). However, Ruangpan et al. (2020) found that only 6% of the

TABLE 1 | Characteristics of the main NbS platforms reviewed in this study, including a short description and purpose of each database and the main features and shortcomings identified, pertinent to this study.

Platform and source	Description and purpose of the platform	Main features	Shortcomings
EcoShape—building with nature	- Consortium formed by 15 parties to promote Building with Nature EcoShape (2020) - Aims to provide guidelines for reproduction of NbS through pilot projects and its respective monitoring results	- Comprehensive content - Data efficiently structure per phase (Overview, Initiation, Planning and design, Construction, Operation and maintenance, Lessons learned)	- Filters are limited to landscapes and technology readiness levels - Lessons learned categories are not standardized between different pilot projects
OPPLA	- Joint output of OPERAs and OpenNESS projects with over 60 contributors OPPLA (2021) - EU collection of NbS case studies - Clear approach to knowledge sharing to a wide audience	- User-friendly interactive map - Standardized structure including available data per section - Keywords inside project's description can be used to search for similar projects	- A few projects provided conflicting information between listed references - Filters limited to scale and type
OURCOAST—ICZM in Europe	- 3-year program dedicated to knowledge sharing around coastal planning and management EC (2012) - Specific focus given to adaptation to climate change, communication systems and planning instruments	- Lessons learned are available - Includes associated costs - Database encompasses a variety of countries - Diversity of police-making initiatives	- Interactive database was discontinued - Access to database seem secluded - Database not very user-friendly
The River Restoration Center (RRC)	- UK's expert center in river restoration, habitat improvement and catchment management the RRC (2014a) - Part of the National River Restoration Inventory (NRRI) the RRC (2014b) - Propagates expertise and provides site-specific technical advice	- Comprehensive structure with standard fields in a PDF format - Contains section dedicated to effectiveness and project's costs - Variety of filters available - Anyone may submit their projects (posted after the RRC's review)	- Lessons learned mostly technical/ engineering-related - Contains only historical data up to 2017 for projects in the UK the RRC (2014c)
RESTORE (RiverWiki)	- Main deliverable of the EU LIFE + RESTORE project the RRC (2014b) - Part of the NRRI the RRC (2014b) - Currently funded by the EA and maintained by the RRC RESTORE (2014)	- User-friendly platform with Wikipedia-like structure - Anyone may submit their projects (posted after the RRC's review) - Global database integrated with the RRC's UK	- Search for case studies might be complicated depending on purpose of search
NATURVATION	- 4-year project focused on building expertise around NbS in urban areas NATURVATION (2017a) - Developed by 14 different institutions in the EU	- Resourceful interactive map - Innovative filters including key challenges, urban setting and project cost NATURVATION (2017b)	- Limited fields and sections - Data was collected between June and August of 2017 and it has not been further updated NATURVATION (2017a)
Climate ADAPT	- Cooperation between the European Commission and the European Environment Agency (EEA) Climate-ADAPT (2020b) - Focused on knowledge sharing about adaptation policies to address climate change-related issues European Environment Agency (2018)	- Complete and standardized project descriptions - Indicates point of contact for each project - Up to date platform - Broad filtering options including type of climate impact and funding Climate-ADAPT (2021a)	- Lack of quantitative data showing effectiveness

analyzed NbS publications between 2007 and 2019 were associated with coastal flooding, and there are even fewer publications presenting data analyzing the success of implemented NbS projects in coastal areas, particularly at a local scale.

The capitalization of results and lessons learnt from previous projects can contribute to overcome key gaps of knowledge and support the replication and improvement of future ecosystem-based projects. This study investigates the application of NbS for coastal climate change adaptation in Europe, based on a detailed review of 59 implemented NbS case studies across European countries. We aim to identify the prevailing characteristics amongst case studies, including the main motivation that triggered the choice of a NbS over a traditional coastal protection approach and the reported co-benefits. Unlike previous reviews, our focus is on case studies information through the review of integrative platforms. Successful examples may include helpful technical details for replication. The analysis leads to the presentation and discussion of identified lessons learned from our selected sample.

TABLE 2 | Design characteristics (left) and classification per characteristic (right).

Design characteristics	Classification
System type	Estuarine; Coastal; River basin
Type of location	Urban; Rural/suburban
Type of infrastructure	Green; Hybrid
Coastal challenge addressed	Reduce flooding; Reduce erosion; Biodiversity restoration/conservation; Reduce flooding and erosion
Type of intervention	Ecosystem creation; Ecosystem restoration; Managed realignment
Ecosystem used	Natural embankments; Wetlands; Salt marshes; Oyster reefs; Beach and dune systems

MATERIALS AND METHODS

Case Studies Compilation

This study analyzed seven platforms (available up to November 2020) that collected information on NbS case studies for climate change hazards in coastal areas in Europe (**Table 1**): EcoShape, OPPLA, OURCOAST, The River Restoration Center, RESTORE (RiverWiki), NATURVATION and Climate-ADAPT. From these sources, we selected case studies that met at least one of the three criteria: 1) the use of ecosystem services was an integral part of the design rationale, 2) it included ecosystem restoration activities such as the removal of engineered solutions or the combination of traditional engineering with the use of ecosystems, and/or 3) it resulted in the creation of new habitats that could provide flood or erosion benefits as well as other ecosystem services. All the case studies we selected were aimed at coastal adaptation to hydrometeorological hazards, either directly (challenge addressed) or indirectly (as a co-benefit). The information from these knowledge sharing platforms was complemented with searches in Google Scholar to gather more information, whenever available, about the case studies listed in the initial search. In total, we collected 59 case studies, which resulted from case studies that met the aforementioned criteria and provided on all relevant design characteristics (**Table 2**). The complete database of case studies is available in the supplementary information (**Supplementary Table S1**).

Variables of Interest

The design characteristics of our case studies selection (n = 59) describe the focus of the project and include system type, type of location, type of infrastructure, coastal challenge addressed, type of intervention and ecosystem used, and each design characteristic contained at least two classes (**Table 2**). The system typologies considered were estuarine, coastal and river basin case studies. The "river basin" type refers to projects that encompass a larger area than the estuarine region and could not be considered "estuarine." Coastal projects cover only the open coastal zone (e.g., beaches and sand dunes systems). The type of locations considered were urban, and non-urban/non-populated, which refers to low-density or uninhabited areas. This classification was based on visual analysis of satellite imagery. The type of infrastructure was classified as: green, when no construction or realignment of engineered coastal defenses is implemented; or hybrid interventions, when ecosystem services were combined with "hard" engineering structures.

The types of funding were classified into public, private, Private-Public Partnership (PPP) or other types of funding. The latest includes trust funds of various structures (e.g., lottery funds, public and private donations), charity contributions, and taxation schemes. In addition, we analyzed the predominant funding sources in the countries where NbS case studies were more frequent. Regarding the information compiled about project monitoring, we registered whether or not monitoring of the case study was conducted. Later, we specifically assessed whether flood and/or erosion effectiveness were indicated by variables such as return periods and accretion rates, respectively.

Information on the "motivation" for the case studies was also revised, especially when the source indicated the motivation for choosing NbS over a traditional approach. The motivation was considered as the goal that would have not been achieved without the NbS component when compared to a traditional solution. For example, if the project goals were environmental compensation and flood protection, the main motivation is registered as environmental compensation because this would not have been achieved by implementing a traditional coastal protection scheme. At least one main motivation was identified per project, and, in the cases where other reasons for choosing NbS were mentioned in the project description as key factors, they were classified as additional reasons. Motivations were grouped in different categories: 1) sustainability, 2) policy-making context, 3) recreation and tourism, 4) cost-benefit relationship, 5) environmental compensation, 6) coastal defense improvement, and 7) development of expertise and knowledge sharing. Each motivation category is described in **Supplementary Table S2**.

Some projects reported co-benefits derived from the implementation of NbS, which were registered and grouped into 13 different categories: 1) biodiversity conservation and restoration, including bird and fish protection; 2) recreation; 3) tourism; 4) reduction of flooding; 5) reduction of erosion; 6) community awareness of coastal and estuarine environments; 7) economic benefits; 8) water quality improvement; 9) educational gains; 10) cost reduction; 11) area availability for housing; 12) navigation; and 13) air quality. Sustainability was considered an intrinsic value to NbS; thus, it was not listed as a co-benefit.

When reported, lessons learned were registered and classified into 10 categories: 1) communication, 2) cost-benefit analysis, 3) funding and costs, 4) planning, design and construction, 5) permitting and legal requirements, 6) biological and ecological, 7) physical, 8) monitoring and maintenance, 9) management, and

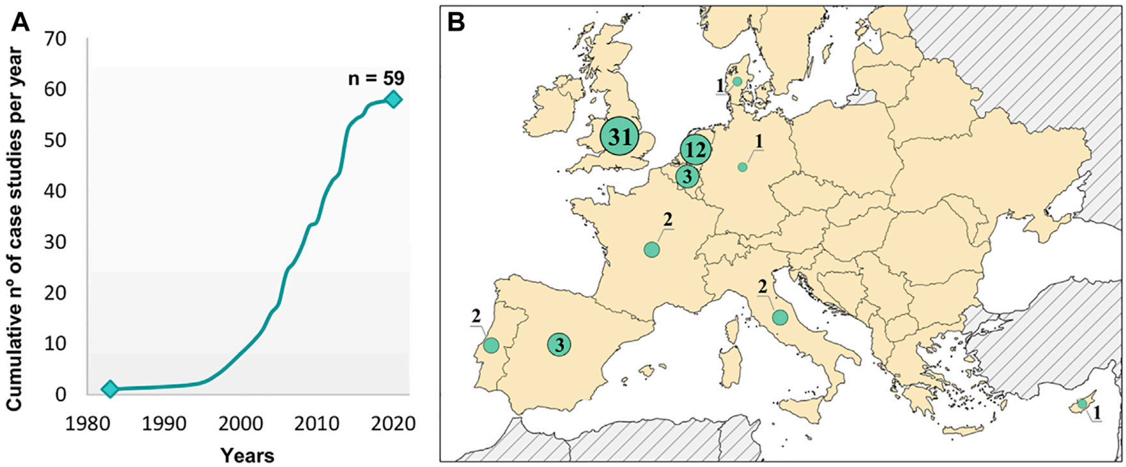

FIGURE 1 | Temporal and spatial distribution of NbS case studies in Europe, shown by **(A)** the cumulative number of projects implemented per year and **(B)** a map of the European continent.

10) stakeholder engagement. A detailed description of each category is presented in **Supplementary Table S3**.

RESULTS

Implementation Status of Coastal Nature-Based Solutions in Europe

From the 59 projects reviewed, most of the projects were implemented from 2002 to present-time; only one was implemented in the 1980s and two in the 1990s. More specifically, the implementation of projects significantly raised between the years 2005 and 2015 (**Figure 1A**). The projects analyzed show a clear regional concentration: nearly 73% (n = 43) of the case studies were located in two countries, the United Kingdom (53%, n = 31) and the Netherlands (20%, n = 12), whereas the other projects were distributed among Belgium and Spain (5% each, n = 3), Portugal, Italy and France (3% each, n = 2), and Germany, Cyprus and Denmark (2% each, n = 1) (**Figure 1B**). Only one of the case studies was a transnational project between Belgium and the Netherlands (not indicated in **Figure 1B**).

Figure 2 provides a summary of the main characteristics of the case studies reviewed. More than half of the projects (69%, n = 41) indicated that coastal protection (reduction of flooding and/or erosion) was the main challenge. Biodiversity restoration and/or conservation was indicated as the main challenge addressed in the remaining 18 projects (31%). More than half of the interventions (61%, n = 36) employed ecosystem restoration, while the other 39% was almost equally split between ecosystem creation (19%, n = 11) and managed realignment (20%, n = 12). Wetlands accounted for 56% of the case studies (n = 33), from which 32% (n = 19) were described as salt marshes and 24% (n = 14) were generally presented as wetlands.

Most of the case studies (64%, n = 38) were considered coastal, and almost one-third (31%, n = 18) were implemented in estuarine areas. Only three cases were described as river basin systems since they cover the estuarine transition to riverine areas, although still tidally-influenced, and are not limited to coastal or estuarine zones (e.g., the "River as Tidal Park" case study in the Netherlands). More than half of the case studies were implemented in urban areas (54%, n = 32) compared to 27 in non-urban areas (46%). The type of infrastructure employed shows a predominance of hybrid solutions (64%, n = 38) over solely nature-based ones (36%, n = 21).

Most of the projects had more than one funding source and it is important to highlight that the same sponsor may have contributed to more than one project (number of contributions exceeds the amount of funding sources). Only 15 projects (25%) reported to have only one source of funding; 36 projects (61%) reported more than one funding source; and eight reported no funding information (14%). It was not possible to identify all the funding sources for each project that indicated more than one sponsor. In total, 130 contributions to NbS case studies were identified from 72 different funding sources. The results indicate a major prevalence of public funding amongst the contributions to case studies (77%, n = 99), while other types of funding represented 17% (n = 22). Only 6% were associated with contributions from private sources (n = 8), and PPP were also infrequent, representing only 1% (n = 1). Other funding mechanisms were only present in the UK and the Netherlands, representing 22% and 8% of the total contributions in each country, respectively. Among public sources, 85% of the projects were funded by local, regional or national governments, whereas only 15% were at least partially sponsored by the European Union through different instruments (e.g., LIFE program, INTERREG program and the Network for Europe grants).

Reportedly, 54 out of 59 case studies were completed or partially implemented, allowing monitoring activities. To assess the monitoring status, the total number of projects considered was n = 54. Amongst them, monitoring was declared in 81% of case studies (n = 44), but more than half did not present any information about flood (57%, n = 31) neither erosion effectiveness (56%, n = 30).

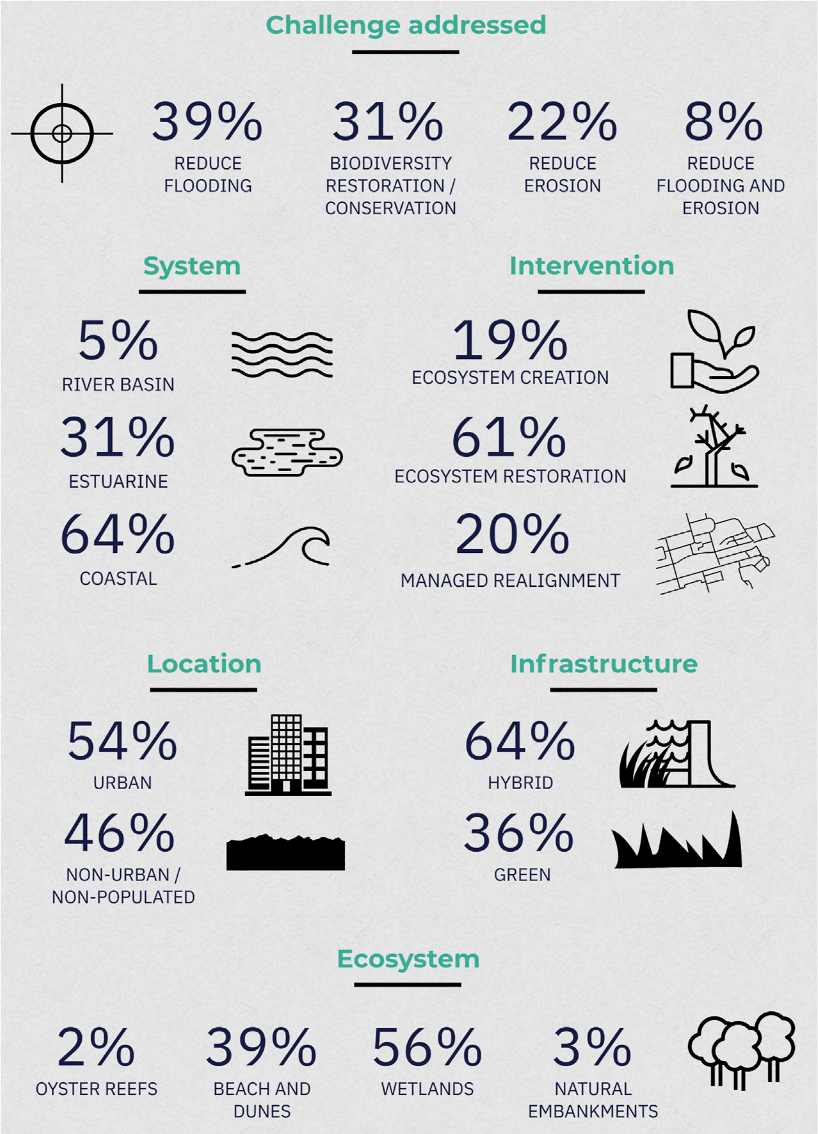

FIGURE 2 | Key characteristics of the review of NbS projects in coastal and estuarine areas in Europe (Infographic produced using Piktochart online tool - https://piktochart.com/).

Additionally, most of the reported monitoring results were related to biodiversity benefits (e.g., presence of water birds). Reported field measurements after implementation accounted for 43% (n = 23) of the case studies. Yet, fewer cases (35%, n = 19) provided detailed results evidencing effectiveness, such as accretion rates or the performance of a flood protection scheme after a storm surge event.

Motivation for Nature-Based Solutions Implementation

The main motivation for the implementation of most NbS case studies (24%, n = 14) was the need for improvement of existing coastal defenses (**Figure 3**). This motivation was also mentioned as an additional reason in 22% of the projects. Developing expertise around NbS implementation and sharing such knowledge ranked second in the case study motivation (19%, n = 11), but it was not mentioned as an additional reason for implementation. "Sustainability" and "environmental compensation" were each cited as the main motivation in 15% of the case studies. "Sustainability" was the most mentioned as an additional reason (39% of the projects examined); however, the need to legally compensate for environmental losses which occurred elsewhere, as defined by Persson (2013), was not mentioned by any of the case studies as an additional reason. Moreover, it was not possible to assert whether when referring to sustainability as a key driver the project owners considered environmental justice and equity aspects as an influential design factor. These aspects should be further explored and

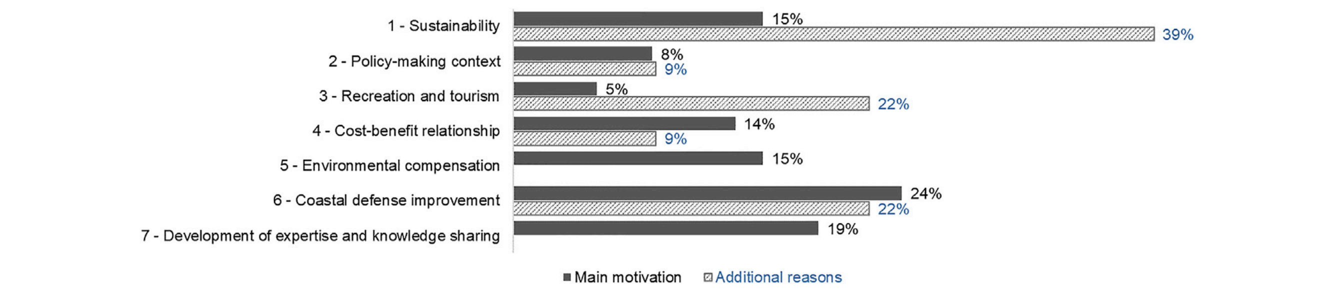

FIGURE 3 | Motivation for the implementation of NbS projects per category in terms of percentage.

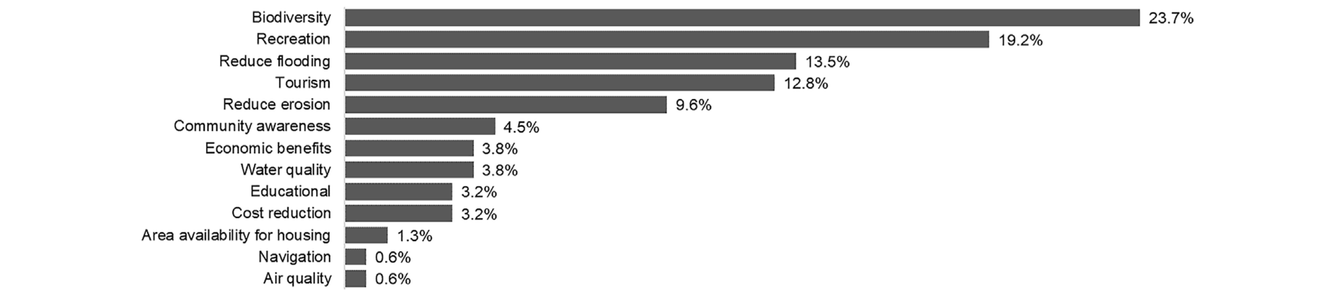

FIGURE 4 | Frequency of co-benefits reported, per category.

explicitly considered in future NbS. Compensation schemes were observed only in three countries: the UK (n = 10), representing 17% of the total number of case studies; the Netherlands (3%, n = 2); and France (2%, n = 1). The influence of the policy-making context was mentioned in 8% and 9% of the projects as main motivation and additional reason, respectively. Recreation and tourism were the least mentioned main motivation (5% of the projects); however, it was mentioned as an additional reason in 22% of projects.

Co-Benefits

A variety of additional benefits (i.e., co-benefits) were reported in the description of the projects. In total, 156 co-benefits were mentioned in the projects examined (**Figure 4**). Four categories of co-benefits represented more than half of the total reported co-benefits (69.2%, n = 108): biodiversity conservation, enhancement and restoration (23.7%, n = 37); recreation (19.2%, n = 30); reduce flooding (13.5%, n = 21); and tourism (12.8%, n = 20). Reduce flooding and reducing erosion (9.6%, n = 15) were considered a co-benefit when the challenge addressed was biodiversity restoration/conservation; only to reduce flooding; or only to reduce erosion. The least mentioned categories included: water quality improvement (3.8%, n = 6); economic benefits (3.8%, n = 6), which were mentioned when there were businesses at risk prior to project implementation; education (i.e., learning outcomes for the community and/or visitors; 3.2%, n = 5); cost reduction in comparison with traditional solutions (3.2%, n = 5); and area availability for the construction of houses (1.3%, n = 2). Some co-benefits were uncommon and very particular to the case study, such as navigation enhancement resulting from sustainable dredging, and air quality improvement from less suspended solids (0.6% each).

Lessons Learned

In total, 155 different lessons learned were reported across all the case studies we reviewed, but many shared common features. Almost one out of four (23.2%, n = 36) of the lessons learned mentioned were associated with the importance of stakeholder engagement (**Figure 5**), including negative experiences of lack of engagement that resulted in project delays. For instance, the creation of a depoldered area between Belgium and the Netherlands as part of the Sigma Plan was delayed due to the opposition of landowners, requiring an improved stakeholder engagement strategy (Climate-ADAPT 2020a). On the other hand, the case study of the shellfish reefs placed in Eastern Scheldt for coastal protection showed the effectiveness of preparing a stakeholder engagement plan which employed a variety of communication methods to reach different interested parties. In 13.5% (n = 21) and 12.3% (n = 19) of the cases, the knowledge on biological and ecological, and physical site-specific aspects, respectively, were mentioned as essential and as a critical gap when unavailable. These lessons learned were generally associated with technical aspects for implementation. For example, the holistic understanding of physical processes affecting sediment transport were a key success factor in the Poole Bay Beach Replenishment Trial (Heron 2016). The relevance of a well-structured

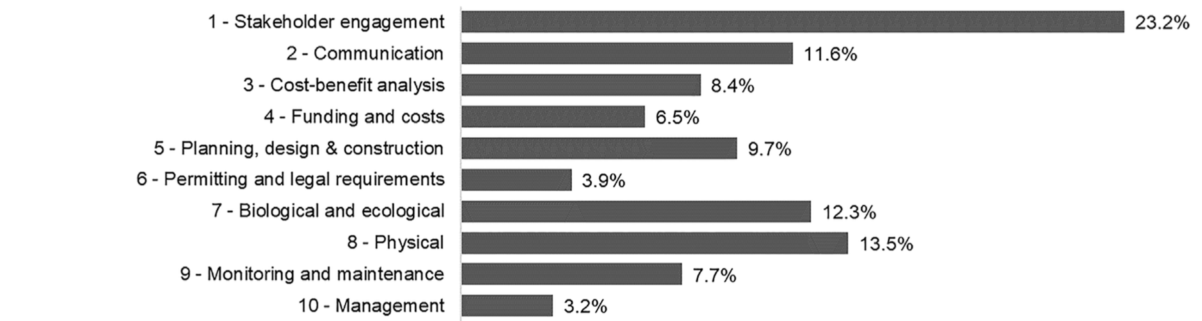

FIGURE 5 | Percentage of reported lessons learned per category defined for the project sample (=59).

communication framework was cited in 11.6% (n = 18) of the lessons learned reports, while optimized planning, design and construction were mentioned in 9.7% (n = 15) of the cases.

The less frequently reported lessons learned included the need for conducting cost-benefit analysis; meeting permitting and legal requirements and timetables; the importance to guarantee funding during the implementation phase; and relevance of resources for monitoring and maintenance (<9% each). The least mentioned was the need for project management experience and multidisciplinary team involved (3.2%, n = 5).

DISCUSSION

This study assesses the status and patterns of implementation of NbS for coastal defense in European countries based on a review of case studies that have been documented and reported in different public platforms in Europe with the aim to support the replication and improvement of future ecosystem-based projects. In comparison to previous reviews on NbS projects, mainly based on scientific literature (Morris et al., 2018; Ruangpan et al., 2020), this study is based on the review of projects implemented, independently of whether the projects delivered scientific outcomes or not, which may not be reflected in scientific articles. We focused on the analysis of a range of project characteristics critical for the implementation of NbS projects in order to support replication. This approach allows a comprehensive assessment on the status of implementation of NbS, not available in scientific studies.

We found that the number of projects implemented over the years has increased during the last decades demonstrating a growing interest in the implementation of NbS for coastal adaptation. In particular, the number of projects increased from 2005 onwards, coinciding with the raise in scientific publications about NbS found by Ruangpan et al., (2020). Yet, an unequal distribution of case studies among countries is perceived in our results, showing that most of the identified projects are located in two countries, the UK (53%) and the Netherlands (20%) whereas all the other eight countries in our database represent less than 5% of the projects. The higher number of NbS case studies in UK and the Netherlands is consistent to their need to respond to already high levels of coastal erosion (Masselink and Russell, 2013) and exposure to flooding (van de Hurk et al., 2006), but it could also be result of further communication, advertisement and reporting of the case studies through regional platforms and other venues (i.e., information inequity). However, it is also likely that these two countries have more experience with coastal NbS given the challenges of flooding in the Netherlands (Jongejan and Maaskant, 2015), and long-established shoreline management plans and wetland compensation schemes in the UK (Doody, 2013). Nevertheless, future scenarios of climate change predict higher increase in the frequency of coastal flooding events in southern European countries (Oppenheimer et al., 2019, as cited in; European Environment Agency, 2019), including Portugal, Spain and Italy, when compared to UK and the Netherlands. These scenarios may justify further expansion of successful cases in southern Europe; yet, according to our results and those of previous studies, experience in southern Europe lags significantly behind in the implementation of coastal NbS.

Considering the substantial investments that the EU has made in NbS research (140 million euros between 2016 and 2017, for instance) (Faivre et al., 2017), reporting on implementation of adaptation measures and their effectiveness for coastal protection are still scarce (Narayan et al., 2016; López-Dóriga et al., 2020; Kumar et al., 2021). Although focusing on green urban infrastructure, Frantzeskaki (2019) has shown that design and scale of NbS directly affect the level of easiness in collecting effectiveness data, which can support replication. Nevertheless, each one of the integrative platforms (**Table 1**) include different information on projects and design characteristics, which limited the identification of projects to a total of 59. This could suggest that some standardization of information from implemented projects, and a joint platform under the EU Commission should help to expand and replicate successful NbS projects. There are also overlaps among the platforms that could be eliminated by integrating different initiatives.

However, replication of these cases should consider local settings and contexts. The effectiveness of NbS is highly associated with the site conditions, which might decrease the relevance of data collected in other locations if they are not carefully assessed and can hamper its replication (Arkema et al., 2017). For instance, water depth, sediment supply, tidal range and vegetation density are factors that affect NbS effectiveness and must be verified in the field as they are site-specific (Pontee et al., 2016). Readily available effectiveness data on implemented small to mid-sized coastal NbS is still scarce, and projects are often lacking on clearly defined baselines that allow a realistic comparison between site conditions before and after the NbS project is executed (Brady and Boda, 2017; Chausson et al., 2020).

Although our results show that most projects are located in coastal areas, there is a lack of scientific papers published on NbS case studies in coastal areas, as indicated by Ruangpan et al. (2020). In addition, our review shows less implementation for some specific ecosystems, for example, oyster reefs compared to coastal wetlands, which is the most applied type of ecosystem. However, most of the coastal wetland NbS are in the UK given the importance of salt marshes in the country associated to habitat losses, land reclamation, coastal squeeze, and the long shoreline management planning that has occurred in the country (Garbutt, 2005; Brady and Boda, 2017; Environment Agency, 2021). Yet, there are also opportunities in other NbS in Europe, such as beach and dune systems (Doody, 2016), and their potential for coastal protection should be further developed and disseminated.

Funding of NbS is largely dominated by public sources with little participation of the private initiative. In the UK, trust funds are a common mechanism used to fund NbS; however, it is unclear why such mechanism is not widespread in the rest of the EU. According to Toxopeus and Polzin (2021), the challenges of combining private and public sources and the scarcity of methods to valuate NbS benefits are the main financial barriers to NbS projects. Whilst there are a number of studies estimating the value of ecosystem-services (e.g., King and Lester, 1995; Barbier et al., 2011; Menéndez et al., 2020; Zhu et al., 2020), economic assessments and financial models are still faulty (Seddon et al., 2020). Furthermore, the COVID-19 pandemic has affected the funding availability for climate adaptation initiatives as governments have committed resources to health and economic incentives (Global Center on Adaptation, 2021). On the other hand, the European Recovery and Resilience Facility will mobilize 724 billion euros to mitigate the economic and social impacts of the pandemic, but also represents an opportunity for a green and sustainable transition (EC 2021c). A minimum of 37% has been estimated for climate investments and reforms, where coastal NbS projects could also be included in support of the implementation of adaptation goals (EC 2021c). In Europe, efforts to tackle climate change challenges are incentivized by different policy frameworks such as the European Green Deal and funding mechanisms such as the EU LIFE sub-program on Climate change mitigation and adaptation. A green recovery highlights the importance of focusing resources on areas that would benefit the most from NbS; however, such mapping of priority areas is still insufficient (Van Coppenolle and Temmerman 2020).

Co-benefits should also be considered while mapping priority areas for NbS implementation. Most co-benefits in this review were related to biodiversity enhancement, which is linked to the most reported additional motivation ("sustainability"). Yet, there is a need to improve the assessment and reporting of other co-benefits, highly relevant for climate change adaptation and mitigation such as flood attenuation, shoreline accretion and carbon sequestration (Nellemann et al., 2009; Duarte et al., 2013). Also, the participation of stakeholders in the identification of co-benefits is vital to support public acceptance (Giordano et al., 2020), as NbS is still seen as an unusual approach by coastal communities (Anderson and Renaud 2021). Emphasis should be given to socio-economic effects arising from NbS implementation and the number of societal challenges that can be addressed by NbS in addition to coastal protection, such as physical and mental health assistance by providing functional green spaces and reduction of unemployment through the creation of green jobs (Davies et al., 2021; Kopsieker et al., 2021).

Based on our results, the case study main motivation was a combination of factors that led to the choice of a NbS project. For instance, the Medmerry Managed Realignment scheme was implemented due to the need to compensate environmental losses elsewhere in an area requiring increased levels of coastal protection; however, traditional structures were no longer financially feasible, and a cost-benefit analysis indicated the ecosystem-based approach as the preferred alternative. Yet, the need of coastal defense improvement was the most mentioned motivation. The explicit inclusion of, quantification of, and monitoring of sustainability is still largely absent from the write-ups and communication of many NbS projects in Europe. Based on this review, the inclusion of measures to monitor and quantify the sustainability of NbS in future projects, for example, through the use of Life Cycle Assessment, aligned with Sustainable Development Goals or the International Organization for Standardization's frameworks for social and environmental sustainability reporting (ISO 2021), should be included to provide metrics on sustainability of the approaches compared to other alternatives.

Our results indicate that hybrid solutions were preferred over the sole use of coastal ecosystems to address efficiency issues of coastal "hard" engineering. Benefits of employing hybrid solutions include reduced maintenance costs of existing structures, increasing the structure's lifespan and avoiding elevated capital costs to build new structures; positive socio-environmental results; and improved coastal protection against flooding (Pontee et al., 2016). The awareness of adverse effects resulting from coastal habitats destruction has increased the use of NbS for coastal protection, although the finer details of the how much protection NbS provides during storms are still insufficient. For example, the role of vegetation in attenuating waves are still not fully understood due to uncertainties associated with vegetation responses during storm conditions and the prediction of vegetation longevity caused by seasonal biomass variations (van Wesenbeeck et al., 2017; Bouma et al., 2014; as cited in; Morris et al., 2018).

The review also highlighted that the reported lessons learned were often poorly described and little detail was given about the implementation experiences. Stakeholder engagement was often mentioned as crucial for the project to guarantee public acceptance, which has been shown to be essential for the mainstreaming of NbS (Anderson and Renaud 2021). As illustrated by the Hesketh Out Marsh Managed Realignment (Climate-ADAPT, 2021b), the engagement of partners is crucial to secure the financial resources required throughout the project, which explains the little amount of lessons learned reports on resources for monitoring and maintenance, assuming that guaranteeing funding for such purposes is considered a component of stakeholder participation. According to our results, vandalism and delays in implementation are amongst the issues caused by the disengagement of the community, which seems to be related to the lack of communication as the importance of a well-developed communication plan was also frequently highlighted. Although technical aspects of each case study might not be as replicable in other locations, sharing knowledge on issues and solutions with the wider public could be essential to help avoid previous mistakes and facilitate replication and implementation of NbS.

CONCLUSION AND RECOMMENDATIONS

As far as the authors are aware, this study is the first that reviews different integrative platforms and compares implemented NbS case studies in Europe, focusing on climate adaptation in coastal and estuarine areas. The study reviews previous NbS projects in Europe to enhance coastal ecosystems conservation and restoration for climate change mitigation and adaptation in coastal areas. The increase of NbS implementation in Europe is highlighted by our results; however its application is still highly biased towards northern countries, and a number of gaps are still hampering the replication of NbS such as general assumptions about NbS effectiveness and lack of site-specific data on physical and ecological processes; the minor participation of the private initiative in NbS funding; and the lack of quantitative effectiveness data in public sources.

Based on this analysis, we recommend that practitioners incorporate a detailed diagnosis of the site prior to NbS implementation in order to effectively evaluate site suitability, need, and community support for such a solution. A clear definition of objectives and expected co-benefits involving as many stakeholders as possible is also recommended. An integration and interconnection of different integrative platforms would be beneficial for the expansion of knowledge sharing networks.

AUTHOR CONTRIBUTIONS

RM, BR, and IM contributed to the conception and design of the study. RM wrote the first draft of the paper. IM wrote the introduction of the paper. IM and MR reviewed all the sections of the paper several times, actively contributing to the development of the final version. All authors contributed to manuscript revision, read, and approved the submitted version.

FUNDING

This research was carried out with the contribution of the LIFE Programme of the European Union to the Project ADAPTA BLUES (ref. LIFE18 CCA/ES/001160).

ACKNOWLEDGMENTS

This document reflects only the author's view, and the Agency/Commission is not responsible for any use that may be made of the information it contains. The financial support to RM during the commencement of this work through an EU-funded scholarship, which was awarded by the Erasmus Mundus Joint Master's Degree in Marine Environment and Resources (MER+), is gratefully acknowledged. BR. acknowledges the support from an Early-Career Research Fellowship from the Gulf Research Program of the National Academies of Sciences, Engineering, and Medicine, although the content is solely responsibility of the authors and does not necessarily represent the official views of the Gulf Research Program of the National Academy of Sciences, Engineering and Medicine. We also thank the reviewers for their important insight and useful comments during the revision of this article.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2022.829526/full#supplementary-material

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ORIGINAL RESEARCH
published: 28 July 2022
doi: 10.3389/fmars.2022.920480

Stability of a Tidal Marsh Under Very High Flow Velocities and Implications for Nature-Based Flood Defense

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OPEN ACCESS

Edited by:
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Specialty section:
This article was submitted to
Ocean Solutions,
a section of the journal
Frontiers in Marine Science

Received: 14 April 2022
Accepted: 03 June 2022
Published: 28 July 2022

Citation:
Schoutens K, Stoorvogel M, van den Berg M, van den Hoven K, Bouma TJ, Aarninkhof S, Herman PMJ, van Loon- Steensma JM, Meire P, Schoelynck J, Peeters P and Temmerman S (2022) Stability of a Tidal Marsh Under Very High Flow Velocities and Implications for Nature-Based Flood Defense. Front. Mar. Sci. 9:920480. doi: 10.3389/fmars.2022.920480

Nature-based strategies, such as wave attenuation by tidal marshes, are increasingly proposed as a complement to mitigate the risks of failure of engineered flood defense structures such as levees. However, recent analysis of historic coastal storms revealed smaller dike breach dimensions if there were natural, high tidal marshes in front of the dikes. Since tidal marshes naturally only experience weak flow velocities (~0-0.3 ms^{-1} during normal spring tides), we lack direct observations on the stability of tidal marsh sediments and vegetation under extreme flow velocities (order of several ms^{-1}) as may occur when a dike behind a marsh breaches. As a first approximation, the stability of a tidal marsh sediment bed and winter-state vegetation under high flow velocities were tested in a flume. Marsh monoliths were excavated from *Phragmites australis* marshes in front of a dike along the Scheldt estuary (Dutch-Belgian border area) and installed in a 10 m long flume test section. Both sediment bed and vegetation responses were quantified over 6 experimental runs under high flow velocities up to 1.75 ms^{-1} and water depth up to 0.35 m for 2 hours. These tests showed that even after a cumulative 12 hours exposure to high flow velocities, erosion was limited to as little as a few millimeters. Manual removal of the aboveground vegetation did not enhance the erosion either. Present findings may be related to the strongly consolidated, clay- and silt-rich sediment and *P. australis* root system in this experiment. During the flow exposure, the *P. australis* stems were strongly bent by the water flow, but the majority of all shoots recovered rapidly when the flow had stopped. Although present results may not be blindly extrapolated to all other marsh types, they do provide a strong first indication that marshes can remain stable under high flow conditions, and confirm the potential of well-developed tidal marshes as a valuable extra natural barrier reducing flood discharges towards the hinterland, following a dike breach. These outcomes promote the consideration to implement tidal marshes as part of the overall flood defense and to rethink dike strengthening in the future.

Keywords: sediment stability, flow velocity, flood risk, dike breach, nature-based adaptation

INTRODUCTION

Low-lying coastal and estuarine areas are increasingly exposed to flood risks as a result of climate change induced sea level rise, increasing storminess, associated storm surges, and land subsidence (Hallegatte et al., 2013; Tessler et al., 2015; Nicholls et al., 2021). Potential impacts in case of floods increase as coastal populations continue to expand (Neumann et al., 2015; Paprotny et al., 2018). This all results in a growing need for climate-resilient flood risk mitigation strategies (Hinkel et al., 2014; Morris et al., 2020; McEvoy et al., 2021). In addition to engineered flood defense structures, such as dikes, the conservation or creation of natural habitats such as tidal marshes and mangroves in front of flood defense structures, can provide additional nature-based flood risk mitigation, by reducing storm impacts on engineered structures (Vuik et al., 2016; Vuik et al., 2018; Zhu et al., 2020a), while at the same time providing ecological benefits such as increased biodiversity, water purification and carbon sequestration (Cheong et al., 2013; Temmerman et al., 2013; Teuchies et al., 2013; Schoonees et al., 2019; Smith et al., 2020). However, uncertainty remains about the functionality of natural habitats as buffers against flood risks under extreme storm conditions.

Relying only on earthen dikes or levees as flood defense structures is risky, as past storm events have shown that dikes can fail and may breach, with dramatic consequences for the communities living in the lowlands behind the dikes. For instance, dike breaching caused the death of more than 1800 people during the North Sea storm in 1953 in the Netherlands (Kabat et al., 2009), more than 1500 deaths due to Hurricane Katrina in 2005 in New Orleans, USA (Day et al., 2007), and displaced more than 100 000 people due to cyclone Aila in 2009 in Bangladesh (Auerbach et al., 2015). Dike breaches result from a structural failure of the dike, i.e., when hydrodynamic forces on the dike exceed the structural strength of the dike. During a storm surge, the hydrodynamic stress generated by high water levels, waves and tidal currents might reach this critical threshold, through mechanisms including dike overtopping by waves or flow, seepage (piping) through the dike, and dike erosion as a result (Vorogushyn et al., 2010; Danka and Zhang, 2015). In NW-Europe, dikes are often constructed of an inner core of non-cohesive sandy material, a top layer of cohesive sediment (i.e. clay or silt), and optionally/often a vegetated cover (Morris et al., 2009; van Loon-Steensma and Schelfhout, 2017). Once an initial disturbance of the top layer reaches the inner sandy core, this non-cohesive sediment will erode more easily, potentially leading to a rapidly expanding dike breach (Visser, 1998; Stanczak and Oumeraci, 2012; Peeters et al., 2015). In many embanked regions the land behind the dikes has a lower elevation compared to the sea or estuarine water level during a storm surge. Due to this elevation difference, a dike breach will result in strong flow velocities and deep flooding into the embanked areas.

In addition to improved response strategies like evacuation, the presence of natural tidal marsh habitats in front of dikes can play a role in mitigating the impacts of dike breaching. Recent analysis of historical dike breach events during the North Sea flood in 1953 in the Netherlands (Zhu et al., 2020a), showed that dike breaches were more narrow and more shallow when tidal marshes were present in front of dikes compared to breaches without tidal marshes in front of them. These findings suggest that tidal marshes serve as an extra natural 'barrier' that restricts the flow discharge towards the dike breach, thereby limiting breach growth and resulting breach width and depth (**Figure 1**). Calculations indicated that the reduced dike breach dimensions behind marshes decrease the flood discharge, and thereby the speed of flooding, the flood depth and hence the potential damage behind the breached dikes (Zhu et al., 2020a). As a result, evacuation procedures will be facilitated. As such, this study showed a new mechanism of nature-based flood risk mitigation by tidal marshes in front of dikes, in addition to the previously shown function of marshes for attenuation of storm waves, currents, surge levels and erosion (e.g. Möller et al., 2014; Spencer et al., 2015; Stark et al., 2015; Carus et al., 2016; Schoutens et al., 2019). Gaining in depth understanding of this new mechanism is highly valuable, as it may inspire novel nature-based flood designs and new integrated flood risk strategies (Zhu et al., 2020a).

However, key questions remain, as there is a lack of direct observations so far on the stability of tidal marshes under the high flow velocities that may be expected over a marsh towards a dike breach (**Figure 1**). In the exceptional case of a dike breach during storm surge conditions, flow velocities over a marsh towards a dike breach may reach up to several ms^{-1}. Direct measurements of such situations are lacking, but estimations for the extreme storm surge and dike breach conditions in 1953 in the Netherlands (**Figure 1**) indicate that the storm surge level was up to 2.6 m above the marsh surface elevation, for which corresponding flow velocities (assuming critical flow conditions) may have reached almost 5 ms^{-1} in dike breaches (based on Zhu et al., 2020a). Flow velocities on a marsh right in front of a dike breach are expected to be lower, due to spreading of the flow over a larger width and due to drag, but may still be in the order of several ms^{-1}. This is much more extreme than the normal tidal conditions under which marsh sediment beds and vegetation naturally develop ~0-0.3 ms^{-1} (Bouma et al., 2005a; Temmerman et al., 2012; Schoutens et al., 2019; Schoutens et al., 2020). Therefore, a crucial question is how stable a marsh can be under such high flow conditions, and hence whether it may serve as an extra natural barrier restricting the flow discharge towards the inundated land behind the dike breach (**Figure 1**). Or in other words, can we rely on the additional strength provided by the tidal marsh to the overall flood defense in reducing the flood risk i.e., preventing or limiting the breach to grow in depth and width?

In general, marsh vegetation and the high intertidal elevation of marshes (i.e. reducing water depth) cause drag to the flow and reduce flow velocities (Carus et al., 2016; Schoutens et al., 2019) and wave heights (Möller et al., 2014; Silinski et al., 2016b; Schoutens et al., 2019). As a consequence of attenuating hydrodynamic forces from waves and currents, tidal marshes have the capacity to trap sediments and organic particles and as such build up elevation and strength (Brooks et al., 2021). Tidal marsh sediments typically have a high fraction of silt and clay particles in combination with a variety of small organic compounds, which increases the sediment cohesiveness

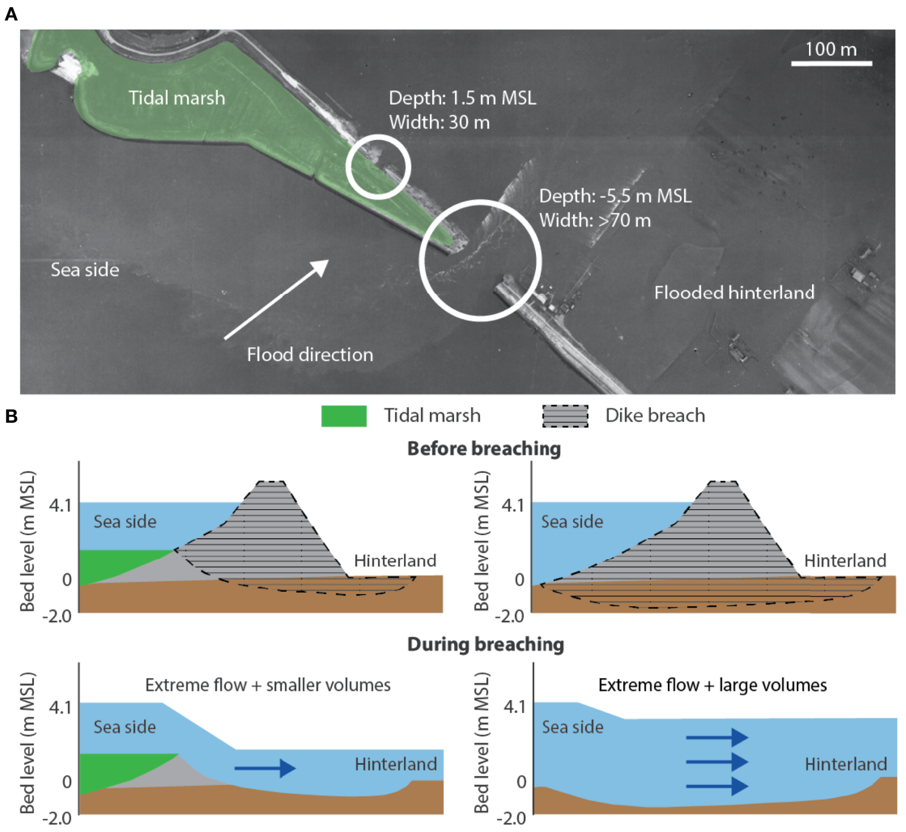

FIGURE 1 | Hypothetical explanation of the protective function of tidal marshes in front of a dike breach. An aerial image of two neighboring dike breaches (white circles) at the former Haringvliet estuary (the Netherlands) during the North Sea flood in 1953 **(A)**. Illustration of how flow velocities and water volume differ in case of a dike breach with tidal marsh (left) and without tidal marsh in front of the dike (right) **(B)**. Credit: Figure adapted from Zhu et al. (2020a).

(Grabowski et al., 2011; Winterwerp et al., 2012). Apart from the small-scale sediment composition, marsh sediments consist of a larger scale network of roots and rhizomes which forms an adhesive between sediments, sediment aggregates and organic compounds (Gyssels et al., 2005; Brooks et al., 2021; Chirol et al., 2021). Belowground plant structures in combination with the cohesive sediments reinforce the structural shear strength of the sediment bed (Shepard et al., 2011; Bouma et al., 2014). Previous flume experiments with simulated storm waves, have confirmed strong resistance of tidal marsh sediments to erosion (Möller et al., 2014; Spencer et al., 2016; Möller et al., 2019). Nevertheless, it is unknown how the tidal marsh vegetation and sediment bed will respond to the high flow conditions during a dike breach event. Moreover, storm surges in NW Europe are typically strongest in the winter season (Masselink et al., 2016; Hansen et al., 2019) when the aboveground plant shoots on tidal marshes die off, and their hydrodynamic attenuation capacity is reduced (Schoutens et al., 2019; Schulze et al., 2019). To understand the effect of high flow velocities on the stability of a tidal marsh with winter-state vegetation, measurements of marsh stability under such conditions are needed.

In this study, we performed flume experiments with tidal marsh monoliths (1.2 m long x 0.8 m wide x 0.4 m high) extracted from the field, exposing them to very high flow velocities in the flume facility to explore the stability of tidal marshes. We studied (1) the resilience of the vegetation in its winter state in combination with (2) the erosion resistance of the sediment. The results of this study will be discussed in light of a new aspect of the nature-based shoreline protection function of tidal marshes, i.e. whether in case of a dike breach, tidal marshes could persist as an extra natural barrier, restricting the flow discharge towards the low land behind the breached dike, mitigate the impacts of a flood and reduce the flood risk.

METHODS

Experimental Setup and Monolith Extraction

This flume experiment was conducted in the Mesodrome flume facility at the University of Antwerp (Belgium) (**Figure 2A**). The flume consists of a 10 m long, 2.0 m wide and up to 1.5 m deep test section and has a maximum pump capacity of 0.6 m^3 s^{-1}. To generate very high flow velocities the width of the test section was reduced to 0.8 m. Within the test section, 8 monoliths were placed to create a marsh of 0.8 m wide and 9.6 m long (**Figure 2B**).

The monoliths were excavated from tidal marshes in the Scheldt estuary (**Figures 3A, B**, 51.35 N, 4.23 E) as sediment blocks of 0.8 m by 1.2 m in surface area and 0.4 m depth, with vegetation growing on top. The monoliths were excavated on January 20 and 21, 2021, from brackish tidal marshes dominated by a mature *Phragmites australis* (Cav.) Trin. Ex Steud. Vegetation (**Figures 3C, D**) which can grow up to 4 meter high (**Supplementary Figure 1**).

Extraction was done by digging a pit, as such creating a vertical cliff in the sediment; then pushing a metal plate horizontally into the cliff at a depth of 0.4 m under the horizontal sediment surface (**Supplementary Figure 2A**); next a rectangular mold of 0.8 by 1.2 m in surface area and 0.4 m high was placed on top of the sediment surface and pushed down gently until it made contact with the metal plate (**Supplementary Figure 2B**). The marsh monolith, contained within a "box" created by the plate and the mold, was lifted and placed on a pallet covered with a thin horizontal, perforated multiplex board (i.e. to support the sediment block, to prevent cracks and to allow a little bit of drainage during the flume experiments). The mold was removed and vertical multiplex boards were attached around the monolith for transport by a truck (**Figure 3D** and **Supplementary Figure 2C**). After placing the monoliths in the flume with a crane, the protective vertical boards were removed from around the monoliths and they were positioned along the 10 m test section (**Supplementary Figure 2D**).

Aboveground biomass of the first 5.6 meters of the monoliths (at the leading edge of the test section) was removed to create a zone without vegetation for the incoming flow before that entered the zone with vegetation remaining along a length of 4 m at the end of the test section (**Figure 2B**). The remaining shoots were cut at a height just below the wooden beams crossing the flume (**Figure 2A**) to exclude interference of bending of the shoots with the beams. The beams were needed to keep the setup in place and withstand the hydrodynamic forces. As such the remaining vegetation stems were max. 1 m high. Small gaps in between the different monoliths and at the side edges were filled with sediment to ensure that the sediment bed was continuous across the flume test section. Before the monoliths were exposed to flow velocities, the flume was filled up with water to the sediment surface to let

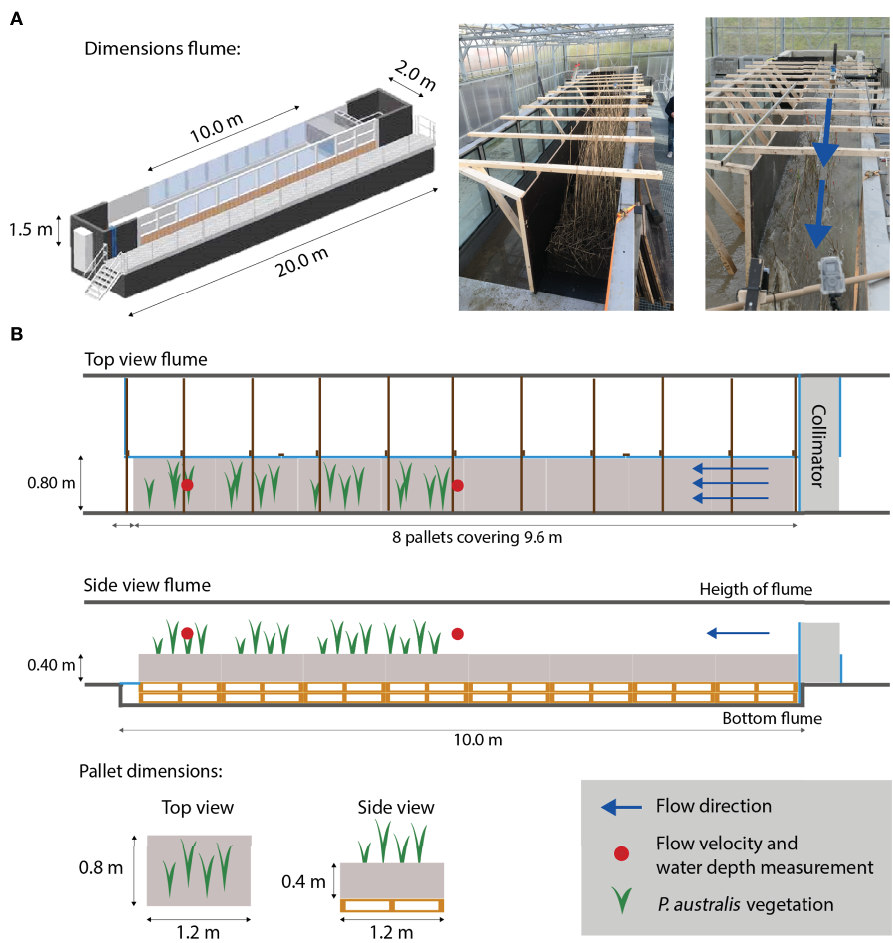

FIGURE 2 | Overview of the flume dimensions and experimental setup **(A)**. Schematic top view and side view of the experimental setup in the flume **(B)**. The flow is generated by a pump that forces water through a collimator into the 0.80 m wide test section. Uniform flow was created by removal of the aboveground vegetation in the first meters of the test section.

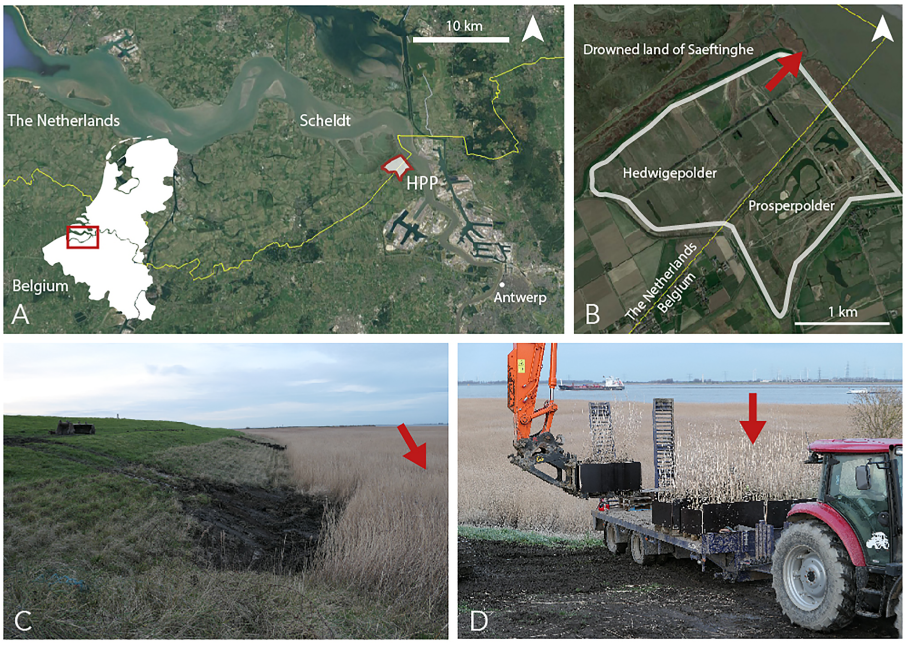

FIGURE 3 | Aerial picture showing the Scheldt estuary from the mouth to the city of Antwerp **(A)**. The monolith extraction took place within the marshes along the Hedwige- Prosperpolder (HPP) at the Dutch-Belgian border (yellow line) **(B)** within a mature, high marsh location adjacent to the dike **(C, D)**. Red arrows indicate the extraction location in the different figures.

the sediment in the monoliths settle for two days. A first set of 8 monoliths was used as a pilot experiment to find out the desired settings (i.e. combination of water level and flume pump rate) to create maximum possible flow velocities with this flume setup. Next, the second set of 8 monoliths was installed in the flume to have undisturbed monoliths before starting the measurements of tidal marsh stability.

Hydrodynamic Flow Conditions

The experiments were conducted as six separate runs of 2 hours. Flow velocity and water depths remained constant during each run, but progressively increased from 1.00 to 1.75 ms^{-1} and from 0.15 to 0.35 m with each new run (**Table 1**). Flow conditions were defined by the Reynolds number (*Re*):

$$Re = \frac{(v_{in}.R)}{v}$$

with v_{in} as depth averaged flow velocity (ms^{-1}), R as hydraulic radius which is defined as the cross-sectional area, A (i.e. for rectangular flumes: $A = w.d$ with w being the flume width which was 0.8 m), divided by the wet perimeter of the flow (i.e. for rectangular flumes: $2d + w$) and v being the kinematic viscosity (10^{-6} m^{2}s^{-1} for water); and the Froude number (*Fr*):

$$Fr = \frac{v_{in}}{(g.D)^{0.5}}$$

with g as the gravitational acceleration (9.81 ms^{-2}) and D as the hydraulic depth which is defined as A/w, which in a rectangular flume is equal to the water depth, d.

The effect of aboveground vegetation cover on the sediment stability was tested by starting with two runs with the original vegetation cover present over 4 m of length of the test section (run 1 and 2), followed by consecutively manual removal (clipping) of 2 m of vegetation (run 3 and 4) and ending without

TABLE 1 | Overview of the six experimental runs with varying vegetation cover (m), water depth (*d*, cm), water surface slope (%), flow velocities (v_{in}, ms^{-1}), Reynolds (*Re*) and Froude (*Fr*) numbers within the vegetation test section.

	Vegetated section (m)	d (cm)	Slope (%)	v_{in} (ms^{-1})	Re ×10^5 (-)	Fr (-)
run 1	4	15	3.3	1.00 ± 0.03	1.09	0.82
run 2	4	25	4.4	1.38 ± 0.04	2.12	0.88
run 3	2	25	3.1	1.41 ± 0.04	2.17	0.90
run 4	2	35	3.3	1.50 ± 0.03	2.80	0.81
run 5	0	25	2.7	1.54 ± 0.05	2.37	0.98
run 6	0	35	3.1	1.75 ± 0.04	3.27	0.94

vegetation cover (run 5 and 6). Flow velocities were measured every run with an electromagnetic flow meter (EMF, Valeport model 801, Totnes, UK) in the middle of the flume width and along a vertical depth gradient with an interval of 5 cm. The flow was measured 4.0 m and 0.8 m before the end of the test section (**Figure 2B**). Flume-wall effects on the flow were regularly checked by expanding the measurement of flow velocities over a cross-sectional grid. At the same positions, water depth was measured to calculate the slope of the water surface.

Characterization of the Monolith Sediment Composition

Site-specific sediment composition was quantified on six surficial sediment samples after removing the top layer of plant litter. Samples from the sediment bed were taken with a Kopecky ring (5.0 cm in diameter and 5.1 cm high and five replicates) and used to determine dry bulk density (after drying at 70°C for 72 h). Next, six mixed scrape samples of the top 2 cm were used to perform volumetric grain size analyses with a Mastersizer 2000 (Malvern) based on laser diffraction after a combined H_2O_2 and HCl treatment to remove organic compounds and disperse aggregates. Organic matter content was determined with the loss on ignition method, i.e., by ashing the samples for 4 hours at 550°C in a muffle furnace (Heiri et al., 2001). Shear strength was estimated based on four replicates with a pocket shear vane tester (Eijkelkamp, NL) for the surface sediment and a field inspection shear vane tester (Eijkelkamp, NL) at 10 cm depth. Penetration resistance of the sediment was measured on four replicates with a penetrologger (Eijkelkamp, NL) with a 1 cm depth interval, and an average of the upper 10 cm was calculated. All samples were taken in close approximation of the monolith extraction site, i.e. within 1-2 meters.

Characterization of the Reed Vegetation

The marsh was covered with a homogenous *P. australis* vegetation in winter-state, i.e. the aboveground biomass consisted of dead, leafless stems, and leaf litter was lying on the sediment bed in between the standing stems (**Supplementary Figure 3**). The reed vegetation was characterized in the field in the same week as the monolith extraction (end of January 2021). Shoot densities were counted at three replicate 0.40 x 0.40 m square plots before all aboveground biomass was harvested and dried at 70°C for 72 h to quantify the aboveground biomass. Shoot lengths and basal shoot diameters were measured on 20 shoots, which were harvested to measure biomechanical properties, i.e. the flexural stiffness and Young's modulus. For the latter, the basal 20 cm of the shoots were used to perform three-point bending tests with a universal testing device (Instron 5942, precision ± 0.5%). For more details on the methods to quantify the biomechanical properties we refer to Schoutens et al. (2021). Belowground biomass was quantified from five replicate sediment cores of 0.10 m diameter sampled up to 0.40 m depth (i.e. the same depth as the monoliths), which were sampled at the location of monolith extraction. The cores were frozen and cut into slices (0-2.5 cm; 2.5-5.0 cm; 5.0-10 cm; 10-20 cm; 20-30 cm; 30-40 cm). For each segment, the sediment was washed out and the remaining belowground biomass was dried (at 70°C for 72 h) and weighed.

Vegetation Response

Within the vegetated test section, 20 shoots were monitored during the first two runs (i.e. 4 meters of vegetation cover). In the third and fourth experimental run, 12 remaining shoots were monitored. In the fifth and sixth run, all vegetation was removed. The bending of the shoots in response to the high flow velocities was quantified in six categories indicating the shoot bending angle compared to the initial situation before the experimental runs. The categories ranged from shoots that did not suffer any damage or reconfiguration (< 5° bending angle) up to heavily bent shoots (> 35°) and broken shoots (i.e. flushed away or clearly broken shoots). The measurements were done at different moments in time, i.e. during the experimental run, directly after the run when the flow was stopped and after one (or three for run 4) day(s) of recovery.

Sediment Bed Response

Bed level changes (by erosion or sedimentation) were quantified by measuring the elevation of the sediment bed before and after every experimental run. Pin measurements [following a Sedimentation Erosion Bar, SEB, approach, see Nolte et al. (2013)] were performed along a grid over the 4 m long and 0.8 m wide vegetated part of the test section with a 5 cm interval in the direction parallel to the flume length and a 10 cm interval perpendicular to the flume length, revealing a total of 444 point measurements.

RESULTS

Properties of the Flow, Sediment Bed and Vegetation

Tidal marsh monoliths were exposed to six consecutive runs of two hours each, with water depths ranging from 0.15 to 0.35 m and depth-averaged flow velocities ranging from approximately 1.00 to 1.75 ms^{-1} (**Table 1**). Flow velocities increased and the water surface slope decreased from runs 1 and 2 (with vegetation over 4 m of the flume length) to runs 3 and 4 (vegetation partially removed and remaining present over 2 m) and runs 5 and 6 (without vegetation cover) (**Table 1**). The combined effect of the flume side walls and the rearrangement of the plant shoots (in response to the flow) towards the center of the flume caused variations of flow velocities over the width of the flume with a standard deviation of 0.1 – 0.2 ms^{-1}. Flow conditions during all runs were estimated to be sub-critical to nearly critical (estimated Froude numbers between 0.81-0.98) and highly turbulent (Reynolds numbers > 10^5) (**Table 1**). The sediment was characterized by a high silt fraction (~72%; 2-63 μm) and clay fraction (~17%; < 2 μm), around 20% of organic matter, and relatively high values of shear strength and penetration resistance (**Tables 2A, B**). *P. australis* vegetation in winter has a more modest aboveground biomass compared to the summer situation i.e., smaller, thinner and more flexible shoots (**Table 3**). The

TABLE 2 | Overview of the sediment characteristics **(A)** at the extraction site presented as the mean, standard deviation (*SD*) and sample size (*N*). **(B)** Field studies with recording of shear vane shear strength measurements near the sediment surface in mature tidal marshes.

a.	Unit	Mean ± SD	N
d_{50}	μm	11.76 ± 0.61	6
clay	%	17.19 ± 1.34	6
silt	%	72.21 ± 2.28	6
sand	%	10.60 ± 2.46	6
Organic matter content (LOI)	%	20.03 ± 0.88	6
Dry bulk density	gcm^{-3}	0.64 ± 0.05	5
Shear vane shear strength at surface	kPa	13.01 ± 4.94	4
Shear vane shear strength at 10 cm depth	kPa	31.75 ± 5.94	4
Penetration resistance (top 10 cm)	kPa	190 ± 40	4

b.

Study	**Shear vane shear strength near surface** (kPa)
Howes et al., 2010	5 – 25
Gillen et al., 2021	16.6
Crooks and Pye, 2000	10.5 – 17.5
Wilson et al., 2012	10 ± 7
Ameen et al., 2017	5 – 20

majority of belowground biomass was found between 10 and 30 cm depth. At a depth below 30 cm, belowground biomass decreased (**Figure 4**). The sediment bed is covered with a layer of litter (e.g. old leaves) under which a superficial network of fine roots can be found (**Figure 8** and **Supplementary Figure 3**).

Response of the Reed Vegetation to High Flow Velocities

Vegetation of *P. australis* in its winter state was able to cope with short-term very high flow velocities. The response of the reed vegetation and the sediment bed dynamics after an experimental run were the cumulative result of all previous experimental runs. During the two hour runs, the shoots were bent heavily in the direction of the flow, but recovered rapidly after the flow was stopped (**Figure 5**). Even after four consecutive runs, total damage remained limited as less than 17% of the sampled shoots were broken and less than 17% were bent more than 35°.

Stability of the Sediment Surface

Over the entire period of the six experimental runs (i.e., 12 hours cumulated exposure time), vertical erosion was limited to a median (± standard deviation) cumulative total erosion of 6.7 ± 2.4 mm only. Over individual runs (2 hours), the median erosion was maximum 2.5 ± 2.5 mm (for run 2) and less than 1.0 mm for all other runs. The first run removed part of the litter and organic debris that was initially covering the sediment surface (**Figure 6** and **Supplementary Figure 3**). In the sections where aboveground vegetation was removed, even more litter was removed in the run directly after removal (i.e. in the 4.0 – 2.0 m and 2.0 – 0.0 m distance from the end of the test section in run 3 and run 5 respectively). The presence or absence of standing *P. australis* shoots had no effect on the bed elevation changes (e.g. ANOVA comparing the vegetated section with the non-vegetated section in run 4: $F_{1,387} = 0.171$, $p = 0.68$) and no systematic spatial patterns were found throughout the six runs (**Figure 6**). Although there is a general trend of slight erosion, at some locations sediment accretion was also observed (**Figures 6**, **7**). Apart from the general trend of limited erosion, outliers of several centimeters of erosion and deposition were observed throughout the entire experiment, as a result of translocated sediment aggregates (**Figure 7**). After the 5th and 6th run, i.e. respectively after 10 h and 12 h of cumulative high flow velocities, first signs of uprooting appeared and revealed a shallow subsurface mat of fine roots (**Figure 8**).

TABLE 3 | Aboveground and belowground properties of reed vegetation (left) at the monolith extraction sites presented as the mean, standard deviation (*SD*) and sample size (*N*) and (right) compared to peak biomass measurements in literature (*mean ± SD*).

	Unit	Mean ± SD	N	Coops et al.1996*	Schulte Ostermann et al., 2021	Zhu et al. 2020b
Shoot density	Shoots m^{-2}	258 ± 28	3	136 ± 13		309 ± 103
Basal shoot diameter	mm	4.6 ± 0.8	20	6.8 ± 0.3	5.3 ± 1.2	5.5 ± 0.6
Shoot length	cm	204 ± 52	20	232 ± 17	241 ± 32	236 ± 62
Shoot mass	g $shoot^{-1}$	4.7 ± 1.3	3	12.2 ± 2.1		
Aboveground biomass	g m^{-2}	1207 ± 346	3			
Belowground biomass	mg cm^{-3}	30 ± 9	5			
Flexural stiffness	Nm^2	0.19 ± 0.15	20	0.55 ± 0.16	0.51 ± 0.41	0.46 ± 0.21
Young's modulus	10^9 Nm^{-2}	6.7 ± 4.3	20	6.9 ± 1.6		9.4 ± 3.8

*(*controlled wave flume experiment).*

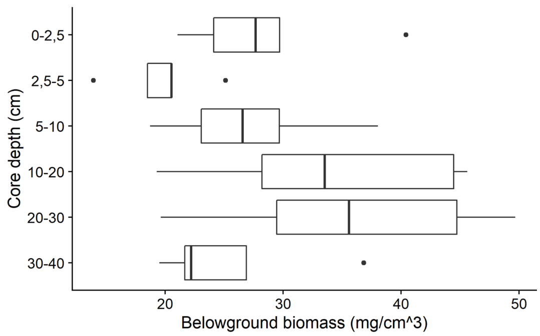

FIGURE 4 | Belowground biomass up to 40 cm depth represented as boxplots with outliers shown as black dots.

DISCUSSION

In face of global climate change, nature-based shoreline and flood protection strategies are increasingly proposed as an adaptive measure to increase the climate-resilience of traditional, engineered flood protection structures. A recent analysis of historic dike breaches highlighted that tidal marshes, when present in front of dikes, can limit the breach-depth in a dike, and hence serve as an extra natural barrier limiting flooding in case of a dike breach (Zhu et al., 2020a). To gain more insights in the actual robustness of these nature-based solutions, we exposed for the first time extended marsh-lengths to very high flow conditions as are likely to happen when a dike breaches behind the marsh. Our flume tests revealed that both the marsh sediment and marsh vegetation show a high resistance against erosion by extended periods of very high flow velocities.

Stability of Tidal Marsh Vegetation in Winter Condition

The results in this study suggest that the reduction of aboveground biomass in winter, reduces the experienced drag which then promotes the resistance of the vegetation against high flow velocities. We tested the winter state stability of *P. australis* vegetation, which is a typical dominant species in the high intertidal zone of brackish tidal marshes in NW European estuaries and in many other brackish and freshwater tidal estuaries worldwide (Srivastava et al., 2014). We found a high capacity of *P. australis* to withstand high flow velocities, as most of the aboveground stems (83%) did not break, of which 66.5% had a less than 25° bending angle at the end of all flume runs (**Figure 5**). At first sight, this finding of high resistance of *P. australis* to high flow velocities may be contrasting with previous studies, showing that *P. australis* vegetation has a lower tolerance to strong hydrodynamic forces, in comparison to the pioneer marsh species that grow on lower intertidal elevations in the brackish parts of NW European estuaries (Coops et al., 1996; Asaeda et al., 2005). Yet, this apparent contradiction between our findings and previous studies may be explained by the following hypotheses. *Firstly*, these previous studies focused on the growth of *P. australis* under average hydrodynamic conditions during a whole growing season instead of short-term extremes in winter conditions. The extreme flow conditions generated on the tidal marsh platform close to a dike breach are expected to last for only a limited time period i.e. usually only one or two high tides that coincide with the storm surge event with a marsh inundation depth of several meters and inundation time of two to four hours per high tide (Stark et al., 2015; Smolders et al., 2020; Zhu et al., 2020a). Once the storm surge has passed, normal tidal conditions prevail again, with very shallow water depths (<0.3 m during high spring tides and even no flooding during neap tides) and weak

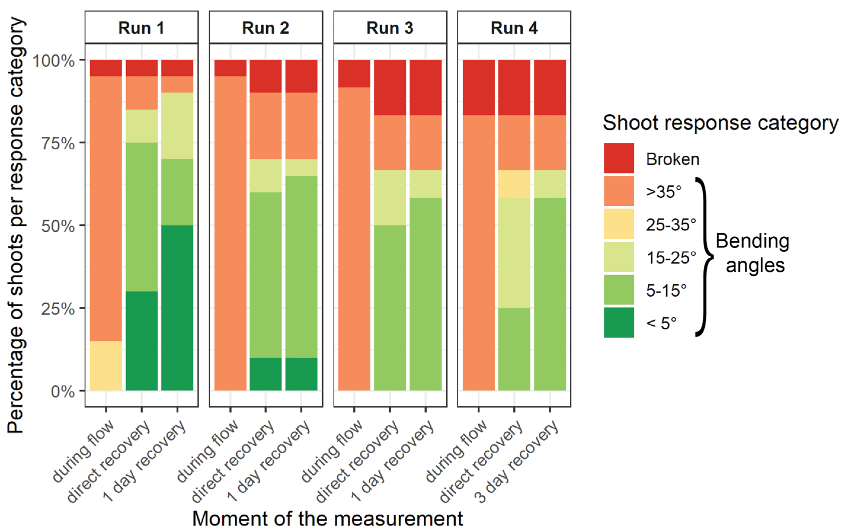

FIGURE 5 | Percentage of the reed shoots (n = 20 for run 1 and 2; n = 12 for run 3 and 4) that show a response in terms of breaking and bending angle when exposed to high flow velocities. The response was quantified in six response categories (broken shoots, and 5 shoot bending angles) and was measured at three moments in time (during the flow, directly after the flow stopped, 1 day later after runs 1-3 and 3 days later after final run 4).

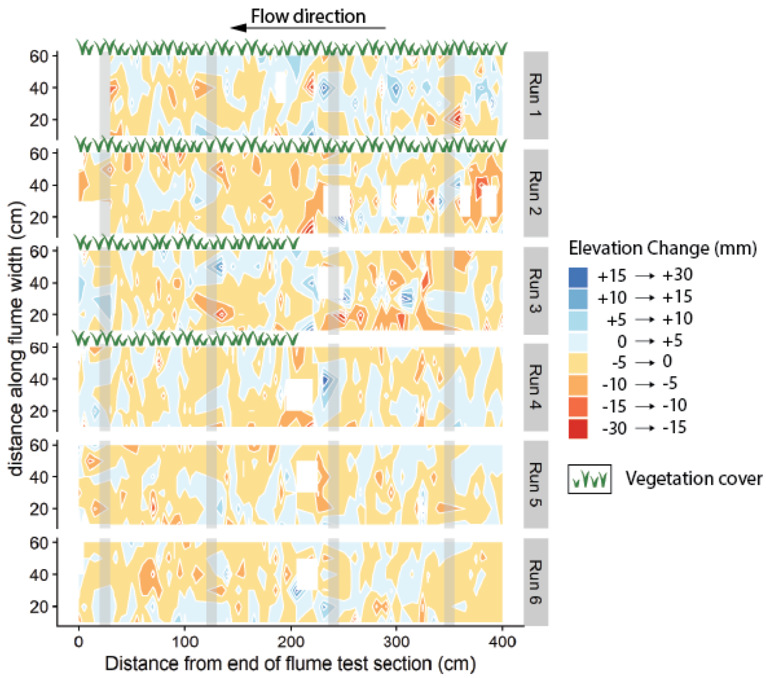

FIGURE 6 | Top view of the flume showing the spatial interpolation of the elevation changes measured during the experimental runs (Run 1 - 6). The elevation changes were calculated as the differences in surface elevation before and after each experimental run. The contours are based on a raster of 444 pin measurements (see methods). White spaces represent missing data. Gray bars represent areas where the space between the pins was 10 cm instead of 5 cm (because of obstruction of the wooden beams supporting the flume construction, see **Figure 2**).

flow velocities (<0.3 ms^{-1}) typically found for the high intertidal marshes investigated here (Bouma et al., 2005a; Temmerman et al., 2012). Hence, the capacity of *P. australis* shoots to cope with high flow velocities is only required for a limited time period in the order of several hours, while previous studies assessed the growth response of *P. australis* to hydrodynamic conditions over a whole growing season (many months) (Coops et al., 1996; Asaeda et al., 2005). *Secondly*, *P. australis* drops its leaves

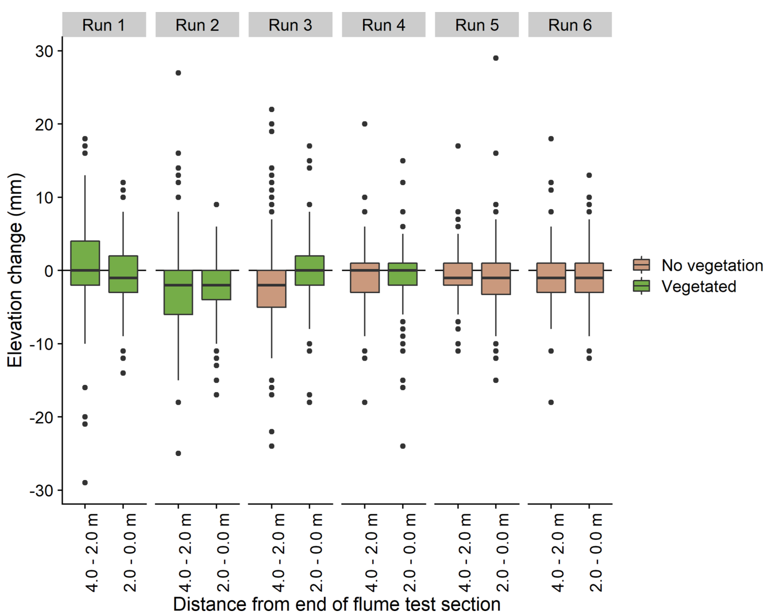

FIGURE 7 | Boxplots summarizing the sediment bed elevation changes as a result of the different experimental runs with outliers represented as black dots. The data were split between the first two meters of the test section (having a vegetation cover in runs 1-2, and no vegetation in runs 3-6) and the back-end two meters (with vegetation in runs 1-4, no vegetation in runs 5-6). Presence or absence of vegetation is indicated by green or brown colors, respectively.

FIGURE 8 | Sediment surface after the last experimental run with the emerging shallow subsurface mat of fine roots. Note that above-ground shoots have been manually clipped.

in winter, which might further contribute to its higher capacity to withstand high flow velocities, as compared to the previous studies on *P. australis* in summer condition. The vulnerability of tidal marsh vegetation to experience stress from hydrodynamic forces is known to be dependent on plant traits that generate high drag forces on the shoots, i.e. plants with high biomass, tall shoots, and stiff stems experience higher drag forces under given hydrodynamic conditions (Bouma et al., 2005b; Schoutens et al., 2020). It should be mentioned that to fit the flume setup of this experiment, *P. australis* shoots were shortened, which could have promoted the resistance during the flow and the recovery process after the flow i.e., shorter shoots experience less drag force during the flow and less downward force during recovery. Nevertheless, compared to values reported in literature for typical *P. australis* summer vegetation, the vegetation from which monoliths were extracted for this experiment had a lower shoot biomass, and stems were thinner and smaller when monolith excavation took place at the start of the winter season in January 2021 (**Table 3**). Moreover, compared to the stiff shoots reported for *P. australis* in summer (**Table 3**), our results suggest that the remaining winter shoots have a higher flexibility. This might be because the stiffest shoots brake and get washed away and only the slightly more flexible shoots remain in winter, which is in line with studies that monitored shoot stiffness of *P. australis* over an entire season (Zhu et al., 2020b).

P. australis winter vegetation in front of a dike can handle short-term high flow velocities, i.e. a cumulative 12 hours in this experiment, without losing much of the shoot biomass (<17% of stems). This finding indicates that after a dike breach has been repaired, the marsh has a high chance to continue providing its shoreline protection function through vegetation-induced attenuation of waves, currents and erosion (Möller et al., 2014; Spencer et al., 2015; Carus et al., 2016; Schoutens et al., 2019; Sheng et al., 2021; Zhang et al., 2022). Furthermore, through survival of the marsh vegetation, the marsh can also sustain its capacity to accumulate suspended sediments that are supplied during regular tidal inundations (Temmerman et al., 2003; Silinski et al., 2016a) and as such, to build up elevation in balance with long-term sea level rise (Temmerman et al., 2004). Studies have demonstrated that tidal marshes in front of dikes, that grow vertically in balance with sea level rise, are very effective in sustaining their nature-based mitigation of waves, and reduction of wave loads on the dikes, under future scenarios of sea level rise (Vuik et al., 2019; Zhu et al., 2020a). Our findings indicate that dike breaching behind marshes, after breaches are repaired, will not compromise this long-term nature-based shoreline protection function of the marshes.

Marsh Sediment Stability Under Short-Term High Flow Velocities

The sediment surface in this experiment was highly resistant against erosion by high flow velocities (**Figures 5**, **6**). Apart from some outliers, the elevation changes ranged predominantly between – 5 and + 5 mm. Here we note that this range of elevation change is not much more than the measurement accuracy of the SEB method (1.5 mm) (van Wijnen and Bakker, 2001; Nolte et al., 2013). Studies on the sediment stability of tidal marshes against vertical erosion under storm surge conditions confirm the highly stable nature of tidal marsh sediments, both found in flume studies mimicking storm conditions (Möller et al., 2014; Spencer et al., 2016) and field assessments after storms (Pennings et al., 2021). Nevertheless, there are also field studies reporting considerable erosion of marshes after very severe storms, such as in freshwater marshes on the Mississippi deltaic plain after the severe 2005 hurricane season (Howes et al., 2010). Moreover, the generated flow velocities in our study go beyond the flow conditions that may be expected on a marsh during a storm surge (Bennett et al., 2020). Nevertheless, the sediment bed remained stable which indicates that the shear strength of the sediment in our experiment was higher than the exerted shear stress (De Smit et al., 2021). In the subsequent paragraphs, we discuss one by one a total of four possible explanations for the observed extremely high erosion resistance of the sediment bed.

Firstly, sediment composition is known to be a key determinant for the susceptibility to erosion of sediment surfaces (Lo et al., 2017; De Battisti et al., 2019; Evans et al., 2021). Thus our finding of high sediment bed resistance to vertical erosion is likely to also be determined by case-specific sediment properties. The grain size distribution of the mineral sediment fraction plays a major role. Erosion resistance of marsh sediments is known to increase with decreasing grain size and associated increasing cohesiveness (Christiansen et al., 2000; Feagin et al., 2009; Lo et al., 2017). Tidal marshes that are situated in front of a dike are often situated relatively high in the tidal frame, being sheltered from the incoming hydrodynamics at the shoreward marsh edge and experiencing relatively shallow flooding at high spring tides, allowing fine sediments to settle. This may explain the high proportions of small sediment fractions, i.e. silt and, to a lesser extent, clay, in the tidal marsh monoliths used in our experiment (**Table 2A**). The high silt and clay content promotes sediment cohesion and is most likely the first reason why our sediment had such a high resistance against erosion.

Secondly, high elevated, mature marshes in front of a dike, like in our case, are typically characterized by strongly consolidated sediments (Tempest et al., 2015), which increases the erosion resistance of the sediment surface (Watts et al., 2003). This is in line with our penetration resistance measurements that were comparable with values found in NW European salt marshes ranging around 200 kPa in the upper sediment layers (e.g. Are et al., 2002; van de Vijsel et al., 2020). Dry bulk densities in this study indicate a soil texture that is favorable for root growth and water drainage (Bradley and Morris, 1990). Although bulk densities are often higher in deeper soil layers, i.e. more compacted, bulk densities in the top few centimeters in this study where in line with values measured in other, natural marshes along NW European marshes, ranging between 0.50 – 0.65 $g.cm^{-3}$ (Crooks and Pye, 2000; Watts et al., 2003; Tempest et al., 2015; Schulte Ostermann et al., 2021).

Thirdly, organic matter content has a positive impact on the erosion resistance of minerogenic tidal marsh sediments and consists of small organic substances and larger belowground root biomass. Our measurements of Loss on Ignition combine these two components and revealed a relatively high fraction of organic matter in the sediments (20.03 ± 0.88%), compared to values in literature ranging between 6-20% for natural mature marshes (Crooks and Pye, 2000; Watts et al., 2003; Tempest et al., 2015; Gillen et al., 2021). In addition to clay particles, small organic substances increase the sediment cohesiveness by forming an adhesive between sediment particles, creating bigger sediment aggregates. Nevertheless, in marshes dominated by organic material (i.e. 80-90% of the sediment fraction), the sediment properties will be different with a lower bulk density and less consolidation (Brooks et al., 2021) which reduces the sediment stability and increases potential erosion processes (Chambers et al., 2019; Himmelstein et al., 2021).

Fourthly, the presence of vegetation can increase the stability of the sediment by (i) reducing hydrodynamic forces due to friction between the aboveground shoots and the moving water and (ii) by providing structural rigidity for sediments and aggregates through the belowground root system (Vannoppen et al., 2015; Cahoon et al., 2020). The sediment surface stability under the short-term high flow velocities in this study was not affected by the presence or absence of aboveground biomass (**Figures 6**, **7**). Rapid removal of the dead organic litter and debris covering the sediment surface was observed after the first experimental runs. In natural marsh conditions, however, the larger scales (i.e., less boundary conditions and larger surface areas) might result in a redistribution of this organic matter rather than complete removal as observed in this flume experiment, hence providing a local shielding function for the sediment surface. Although logistical restrictions did not allow to install fresh, undisturbed monoliths for every single run, the elevation changes in run 2-6 might be more similar to the observations after run 1 in which there was a redistribution of organic matter rather than a removal. During the high flow velocities, *P. australis* shoots were completely bent over (**Figure 5**), hence the friction with the water column was reduced. Although not directly tested, the high fraction of organic matter and the high portion of root biomass at the sediment surface might have had an important contribution to the sediment stability. That is, a high fraction of root biomass near the sediment surface (**Figure 4**) might cause a decrease in bulk density by creating pores and voids between the sediment particles (Brain et al., 2012; Chen et al., 2012; Jafari et al., 2019). Root networks change the sediment characteristics both through the presence of dead and living roots which then function as a structural framework for the sediment aggregates, promote the formation of pores and enhance drainage capacity, hence increasing the erosion resistance of the sediment surface (Gyssels et al., 2005; Grabowski et al., 2011; Evans et al., 2021). Only in the two highest flow conditions (after 10-12 h of cumulative flow exposure), uprooting of fine roots covering the entire sediment surface was observed (**Figure 8**). In addition, when roots get exposed at the sediment surface, they will cause local turbulence, which may result in scour features (Bouma et al., 2009; Schoutens et al., 2021). This might suggest that when high flow velocities would continue for a much longer period of time, the top layer of sediment may get damaged or removed, exposing the subsurface sediment layers with lower belowground biomass. Moreover, vertical variations in the shape of the roots, e.g. from dense fine roots in the upper sediment layers towards sparser thicker roots in deeper layers (Gillen et al., 2021) or reduced belowground biomass with increasing depth (Howes et al., 2010), can alter the shear strength too. Nevertheless, subsurface shear vane tests indicated highly stable sediments limiting the erosion risk (**Table 2A**), even when the top layer of sediment including the dense root network is gone.

Suggestions for Further Research

Despite the fact that we simulated maximal, near to critical flow conditions in our flume experiment, with depth averaged flow velocities up to 1.75 ms^{-1} (**Table 1**), we recognize that the water depths in our experiment were, for practical reasons, limited to maximum 0.35 m. Under extreme storm surge conditions, when water depths on marshes can be as much as 1.5 to 2.5 m (e.g. Stark et al., 2015; Zhu et al., 2020a), it may be well possible that maximum flow velocities over a marsh nearby a dike breach reach up to several ms^{-1}. Therefore, the present flume experiment needs to be regarded as a first test, which reveals promising results on tidal marsh stability and motivates further testing under larger water depths and higher flow velocities.

Current findings are based on one specific tidal marsh characterized by fine, cohesive sediments with a high shear strength and a monospecific *P. australis* vegetation. Although we argue that there are marshes with similar characteristics, we recognize that many other types of vegetated marshes exist e.g., minerogenic and organogenic marshes. Although this study provides insights in the stability of tidal marshes under high flow velocities, further research would be needed to confirm our findings for a wider range of hydrodynamic conditions and variety of marsh types.

Further research should focus on increasing the water depth and the flow velocities to simulate more extreme storm surge conditions. Field experiments with in situ, controlled dike breaches could be an option, however they remain logistically challenging (Peeters et al., 2019, Peeters et al., 2015; Wu et al., 2011). Furthermore, such controlled dike breach experiments are usually conducted during calm or moderate weather conditions and during normal tidal conditions, when water depths and flow velocities over marshes are small compared to extreme storm surge conditions. Finding suitable test locations with low elevated marshes in front of a dike is challenging, as the opportunities for *in situ* dike breach experiments are very limited. Lowering the marsh platform or lowering the dike could be an option to simulate the water depths and flow velocities expected during a dike breach. Experiments in large and deep flume facilities could be an alternative to control the water depth and flow velocity without taking into account the natural tidal cycling.

Apart from experiments with more extreme flow conditions, exploring the sediment stability of different types of vegetated marshes is advised. Several studies point towards the role of plant traits in stabilizing the sediment based on the structure of the root network (Howes et al., 2010; Jafari et al., 2019; Gillen et al., 2021). *In this study, P. australis* formed dense, shallow mats of roots which could benefit the sediment stability compared to species that only develop roots in the deeper sediment layers, suggesting that the type of roots and their structure might play an important role in the sediment stability (Feagin et al., 2009; De Battisti et al., 2019). Other studies emphasize the effect of sediment composition, such as the decreasing sediment stability in function of an increasing fraction of organic substances (Gillen et al., 2021) or an increasing fraction of coarser, sandy sediments (Schoutens et al., 2019; Evans et al., 2021). Moreover, climate induced environmental changes might alter the stability of tidal marshes, i.e. increased stress from inundation by sea level rise and increased hydrodynamic forces from storms could alter the sediment dynamics (Schuerch et al., 2018; Jiang et al., 2020) and the growth of tidal marsh vegetation (Kirwan and Guntenspergen, 2012). Therefore, testing the sediment stability and stabilizing effect of different vegetation types in different environmental settings (e.g. climate) will contribute to the generality of our findings and could improve our understanding of how the stability of vegetated marshes under extreme flow velocities could change in the future. In the first place, we advise to test vegetated marshes which typically occur in front of dikes, such as salt marsh species (e.g. *Elymus* sp. or *Spartina* sp.), freshwater marsh species and tropical systems such as mangroves.

In the present study, we focus on a mature, high elevated marsh adjacent to the dike. However, the response to extreme flow velocities might be different in marshes with a more complex geomorphology created by features such as creeks (Symonds and Collins, 2004). Moreover, young, low-lying developing marshes are expected to have different, less stable sediment characteristics (Evans et al., 2021), which is especially the case when tidal marshes are (re)created, e.g. in managed realignment projects (Tempest et al., 2015; Van Putte et al., 2020). Implementations of nature-based shoreline protection by tidal marshes has increased over the past decades and will further increase in the future. Many projects involve new marsh development on low elevated, initially bare flats in front of the embankment, where sediment accretion might be relatively fast (Vandenbruwaene et al., 2011; Oosterlee et al., 2018), resulting in less consolidated sediments, low bulk densities, poor drainage and hence lower shear strengths (Watts et al., 2003; Van Putte et al., 2020). Experience from existing tidal marsh (re)creation projects suggests that it might take several years before marshes develop with sediments that have enough strength to withstand high shear stresses (Fearnley, 2008; Kadiri et al., 2011; Tempest et al., 2015; Sha et al., 2018) as can be expected on the marsh platform in front of a dike breach. In this respect, further research is needed on the rates of development of marsh sediment strength, as marshes develop from young pioneer marshes to older established marshes.

Implications for Flood Risk Management

Our results indicate that conservation, restoration and creation of tidal marshes in front of engineered flood protection structures such as dikes, provides an extra natural 'barrier' that can remain stable under high flow velocities, and as such may reduce flood depths and damage in case of breaching of the engineered dike. Note that this extra barrier-function requires stable high marshes which need time to develop, but which have the capacity to adapt and develop with changing environmental conditions such as sea level rise. Hence, instead of constructing dikes directly adjacent to the mean low water level, thereby embanking and losing pre-existing tidal marshes and diminishing the chance of new seaward tidal marsh establishment, we stress the benefits of providing enough space for tidal marshes in front of dikes. In areas where coastal development by building of dikes is ongoing, such as in Tianjin (China), Jakarta (Indonesia), or Port Harcourt (Nigeria) (Martín-Antón et al., 2016; Sengupta et al., 2018; Zhang et al., 2020), land reclamation often leads to degradation of many of the ecosystem functions and can bring extra costs that are often not accounted for in the reclamation project (Wang et al., 2010; Tan et al., 2020). Therefore, preserving part of the tidal marshes in front of newly constructed dikes is strongly advised.

In cases where no tidal marshes are present, the creation of suitable conditions for marsh development is recommended (Van Loon-steensma and Slim, 2013). At locations with enough space shoreward of the dikes (e.g. along shallow bays or lagoons), tidal marsh establishment can be facilitated by creating shallow, sheltered conditions by sediment nourishment and building (temporal) barriers to reduce hydrodynamics and capture sediments (Hofstede, 2003; Dao et al., 2018; Vuik et al., 2019; Baptist et al., 2021; Hu et al., 2021). Furthermore, active planting of marsh plants may eventually further enhance the chances of tidal marsh establishment (Tagliapietra et al., 2018).

At locations where space is limited shoreward of dikes (e.g. embanked estuaries with narrow shipping channels), it is worth to explore areas where there is space for more landward building of a new dike and breaching of the existing dike, so-called managed coastal realignment. Managed realignment creates a sheltered environment, suitable for tidal marsh development between the breached and the new dike. This managed realignment strategy has been implemented in several places over the last decades (Esteves and Williams, 2017; van den Hoven et al., 2022), for instance in Belgium where a total of 2500 ha of tidal marshes will be (re)created by 2030 (Temmerman et al., 2013). In places where managed realignment is not a desirable option, one could think about double-dike systems with transitional polders (i.e., see **Figure 7** in Zhu et al., 2020). This implies that after a high marsh has established, the land can be converted back to its former (often agricultural) function, while still offering the benefits of an elevated foreshore. However, as stated before, it is important to account for the time needed to develop stable marsh sediments. This means that if we think nature-based solutions are a promising option to cope with sea-level rise, we must plan well in advance in order to initiate the development of new marshes as part of any kind of nature-based flood protection program.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

AUTHOR CONTRIBUTIONS

KS: Preparation of the experiment, performing the measurements, data analysis and visualization, interpretation of results, writing - review & editing. MS: Performing the measurements, data analysis, writing. MB: Experimental design, interpretation of results, review & editing. KH: Experimental design, interpretation of results, review & editing. TB: Provisioning of facilities, experimental design, performing the measurements, interpretation of results, review & editing. SA: Experimental design. PH: Experimental design. JL-S: Experimental design, interpretation of results, writing - review & editing. PM: Provisioning of facilities, experimental design. JS: Provisioning of facilities, preparation of the experiment, experimental design, interpretation of results, writing - review & editing, PP: Provisioning of facilities, preparation of the experiment, experimental design, interpretation of results, writing - review & editing. ST: Provisioning of facilities, preparation of the experiment, experimental design, interpretation of results, writing - review & editing, supervision. All authors contributed to the article and approved the submitted version.

FUNDING

The Mesodrome flume facility was funded through AnaEE-Flanders projects I001921N and G0H4117N. We would like to thank the funders of this research, i.e. Interreg Polder2C's consortium funding the Living lab Hedwige-Prosperpolder project and personal funding from the Research Foundation Flanders, Belgium (FWO, PhD fellowship for fundamental research K. Schoutens, 1116319 N). JS is grateful to the Antwerp University Research Fund (BOF; Project no. 43171). SM, MB, KV, TB, SA., PH, JL-S. are part of the project The Hedwige-Prosper Polder as a future-oriented experiment in managed realignment: integrating saltmarshes in water safety (with project number 17589) of the research programme 'Living Labs in the Dutch Delta' which is (partly) financed by the Dutch Research Council (NWO).

ACKNOWLEDGMENTS

Thank you Geert Vanwesenbeeck (Jan De Nul NV) and Baecke B.V. for the logistical support in excavating the monoliths and placing them safely in the Mesodrome flume facility. We thank Lennart Van IJzerloo for performing the biomechanical measurements at NIOZ, Yerseke. Thank you to all volunteers and especially Heleen Keirsebelik, Ignace Pelckmans and Dimitri Van Pelt for the logistical support by preparing the flume set-up.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2022.920480/full#supplementary-material

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frontiers | Frontiers in Ecology and Evolution

TYPE Original Research
PUBLISHED 07 September 2022
DOI 10.3389/fevo.2022.699201

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Ecosystem-based disaster risk reduction can benefit biodiversity conservation in a Japanese agricultural landscape

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Ecosystem-based disaster risk reduction (Eco-DRR) has attracted increased attention as a sustainable way to achieve both disaster risk reduction and biodiversity conservation, although there have been few quantitative evaluations of the potential impacts of Eco-DRR on biodiversity. Here, we examined the influences of flood hazard and land-use patterns on biodiversity by focusing on the species richness of plants, butterflies and odonates, and the abundance of two frog species in a rural landscape of Wakasa town, Fukui Prefecture, Japan. The direct effect of exposure to flood hazard on the studied taxa was not significant, whereas landscape factors associated with flood hazard significantly influenced either of the taxa in different magnitudes. We then exercised a scenario analysis by replacing urban land-use by non-urban, agricultural land-use (paddy fields in this case) to reduce exposure to flood hazard and projected the impacts on biodiversity. Our results demonstrated that the land-use replacement potentially reduces the risk of flooding by up to 5.19 billion yen (ca. 46 million US$) and, at the same time, positively influences the species richness and abundance, although the ecological impacts are different depending on taxon and spatial location. The land-use replacement was expected to result in the increase of plant richness and abundance of Daruma pond frog at a location by up to 16 and 25%, respectively. On the other hand, butterfly richness at a location was presumed to decrease by until −68%, probably due to their dependence on domestic gardens. The abundance of Japanese wrinkled frog did not show such a clear spatial variation. This study highlights the significance of land-use replacement as an Eco-DRR measure to reduce the disaster risk and conserve biodiversity in the agricultural landscape.

KEYWORDS

biodiversity conservation, disaster risk reduction, paddy fields, floodplain, scenario analysis, Eco-DRR

OPEN ACCESS

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SPECIALTY SECTION
This article was submitted to Interdisciplinary Climate Studies, a section of the journal Frontiers in Ecology and Evolution

RECEIVED 23 April 2021
ACCEPTED 22 August 2022
PUBLISHED 07 September 2022

CITATION
Kasada M, Uchida K, Shinohara N and Yoshida T (2022) Ecosystem-based disaster risk reduction can benefit biodiversity conservation in a Japanese agricultural landscape.
Front. Ecol. Evol. 10:699201.
doi: 10.3389/fevo.2022.699201

Introduction

Disasters caused by natural hazards have been increasing over the past decades because of urbanization, intensified land use, change of environmental quality and climate change (e.g., Peduzzi et al., 2009; Cui et al., 2019; Williams et al., 2019; Ward et al., 2020). Flooding causes one of the most lethal and economic damages and warrants special consideration (e.g., Degiorgis et al., 2012; Kazakis et al., 2015; Khajehei et al., 2020), so that flood risk management set to be a principal issue frequently in social and economic planning worldwide. Recently much attention has been paid to ecosystem-based disaster risk reduction (Eco-DRR) as an alternative countermeasure for naturel disasters. Disaster risk is usually expressed as the interaction between hazard, exposure and vulnerability (UNISDR, 2009). Eco-DRR utilizes various functions of ecosystems for reducing the risk of disaster events by lowering either or combination of hazard, exposure and vulnerability, as well as for sustainably managing and protecting natural resources provided by ecosystems and biodiversity (Cohen-Shacham et al., 2016; Renaud et al., 2016; Sebesvari et al., 2019). Eco-DRR is defined as "the sustainable management, conservation and restoration of ecosystems to reduce disaster risk, with the aim to achieve sustainable and resilient development" (Estrella and Saalismaa, 2013). Evaluation of the effectiveness of Eco-DRR in various aspects becomes important more and more these days as the demands and attempts of Eco-DRR have been increasing, although there still exist research gaps in various aspects of Eco-DRR in different regions of the world (Sudmeier-Rieux et al., 2021).

The effectiveness of Eco-DRR has been evaluated in previous studies from the points of biophysical, economic, and water quantitative and qualitative impacts, and the advantages of Eco-DRR over other measures have been shown in some cases of previous studies. For example, the Low Impact Development that is an approach to control storm water through the creation of a landscape fostering a natural hydrologic regime has been shown to reduce runoff of surface water by up to 70% and pollutant concentrations by up to >95% (Dietz, 2007; Dietz and Clausen, 2008). Also, floodplain preservation can generate economic benefits to avoid flood damages to property, counting 2.6 million US$ in the case of the East River Watershed, Wisconsin, United States (Kousky et al., 2013) and 7.7 million US$ in St. Louis County, Missouri, United States (Kousky and Walls, 2014). In addition to these advantages, Eco-DRR is expected to give positive effects on ecosystems and biodiversity (Cohen-Shacham et al., 2016; Renaud et al., 2016). Conventional hard-engineering measures of disaster risk reduction could impact ecosystems and biodiversity negatively, but this negative impact can be improved if a win-win relationship between biodiversity and human well-being such as disaster risk reduction is achieved (e.g., Kasada et al., 2017). In fact, coexistence of restoring nature and flood risk control can be achieved and contribute to sustainable countermeasures against flood (Mah, 2011). Areas exposed to higher flooding hazard are considered to experience more frequent flooding, and such areas often used to be wetlands (e.g., riverine floodplains) that frequently harbor high biodiversity (Gibbs, 2000; Tockner and Stanford, 2002). Thus, Eco-DRR that promotes the use of wetlands and floodplain has the potential to contribute to biodiversity conservation. Furthermore, various ecosystem services including provisioning, regulating and cultural ones are also expected to be provided by Eco-DRR (e.g., Cohen-Shacham et al., 2016; Huang et al., 2021).

In spite of the potential contribution of Eco-DRR to biodiversity conservation, there have been limited quantitative evaluations of the impacts of Eco-DRR on biodiversity (IPBES, 2019; Seddon et al., 2020). This may be due to the complex nature of Eco-DRR in how ecosystem-based approaches to reduce risk components (hazard, exposure and vulnerability) can benefit biodiversity conservation. For example, in the case of riverine flood, conserving natural floodplain habitats lowers the exposure of human lives and properties to flood and contributes to biodiversity conservation, whereas flood hazard, which can be reduced by storing storm water in wetlands, also influences biodiversity as it is associated with disturbances that have significant impacts on ecological communities (e.g., Townsent et al., 1997; Pollock et al., 1998). Previous studies have mainly focused on the aspect of habitat conservation or creation of Eco-DRR and revealed the positive impacts on biodiversity (e.g., Liquete et al., 2016; Renaud et al., 2016; Nakamura et al., 2020), although the impacts of Eco-DRR on biodiversity would include other aspects such as the direct effects of hazard on populations and communities of organisms (i.e., disturbance as an ecological process) and the landscape factors that are relevant to animal behavior or meta-population/community, which would be dependent on the exposure to hazard. Then, we hypothesized that the Eco-DRR approach provides overall positive effects on local biodiversity while reducing the potential flood risk, although the effects on local biodiversity would be different depending on species life history and spatial variations in other environmental conditions. This study focused on the direct effects of flood hazard as disturbance and the landscape factors on biodiversity in order to contribute to the understanding of impacts of Eco-DRR on biodiversity.

In this study, we first examined how flood hazard and land-use patterns influence biodiversity independently and interactively in a Japanese rural landscape that is predominated by paddy fields. Rice paddy fields provide spawning and nursery grounds for diverse organisms and thus function as alternative habitats to natural wetlands (Fasola and Ruíz, 1996; Elphick, 2000; Kano et al., 2010; Sesser et al., 2018; Kasahara et al., 2020), although the function is subject to actual agricultural practices (Lawler, 2001; Machado and Maltchik, 2010; Katayama et al., 2011, 2015). We also examined biodiversity in semi-natural grasslands that are formed on the levees in between paddy fields and maintained by periodic mowing (Shinohara et al., 2019). Thus, our studied organisms included plant, butterfly, odonate and frog species that frequently occur in the rural landscape consisting of paddy fields and the associated semi-natural grasslands, although the response to flood hazard and land-use patterns would be different depending on the taxa.

Secondly, we conducted a scenario analysis based on the results of the abovementioned analysis. Scenario analysis can play a significant role in better informing local stakeholders and decision makers (IPBES, 2016), which is an important step to promote the implementation of Eco-DRR on the ground. In our scenario, replacement of urban land-use to agricultural land-use (paddy fields) is considered as a potential measure to reduce the exposure of properties to flood hazard and thus lower the potential flood damages as a measure of Eco-DRR (e.g., Nishihiro et al., 2020; Osawa et al., 2021). Although flood in paddy fields potentially causes loss of agricultural production and damages of agricultural infrastructure, those economic loss is considered to be much less compared to potential loss and damages in urban land-use (Ministry of Land, Infrastructure, Transport and Tourism, Japan [MLIT], 2020a), resulting in the significant reduction of flood risk by the replacement of urban land-use to paddy fields in flood hazard area. Also, land-use planning to reduce exposure to flood hazard is a part of the new national policy of flood risk reduction (Ministry of Land, Infrastructure, Transport and Tourism, Japan [MLIT], 2020b), so that our scenario analysis would contribute to the understanding of consequences of the policy in the studied area. We examined two types of scenarios that assumed population decline or constant population, which was represented as the conversion of urban land-use with agricultural one or the swapping of urban and agricultural land-use, respectively (see details in methods). Replacement of urban land-use to natural floodplain would be ideal in terms of biodiversity conservation, although it might be not realistic and feasible in the actual rural community. Thus, we consider replacement to agricultural land-use (paddy fields) as an alternative. We thus considered two kinds of scenarios and projected the impacts on biodiversity and the risk of flood hazard in the studied rural landscape, although plausibility of the scenarios is a matter of future research. The first scenario is replacement of urban land-use with flood hazard to agricultural land-use to reduce the exposure and damage from flood, called "conversion scenario." The second one is replacement of urban land-use with flood hazard to agricultural land-use and, in turn, replacement of agricultural land-use without flood hazard to urban land-use. This results in swapping between urban land-use with flood hazard and agricultural one without hazard, called "swapping scenario."

Materials and methods

Study area, flood hazard and land-use

We studied an area that used to be a floodplain and currently dominated by paddy fields in the southwest part of Fukui Prefecture, western Japan (8 × 10 km; 35°33′N, 135°54′E; **Figure 1**). We established 72 sampling sites for the biodiversity survey on the basin of the Wakasa-region (**Figure 1A**). These sites were chosen so that they distributed as uniformly as possible in the whole study area. The mean annual temperature was 15.1°C, and the mean annual precipitation was 2,229 mm (2000–2020), which were recorded in a nearby automated meteorological data acquisition point (35°36′N, 135°55′E) according to the Japan Meteorological Agency (2021). In this region, farmers plant rice in paddy fields once a year, and levees between paddy fields are managed as semi-natural grasslands that are maintained by periodic mowing, which serve as important habitats for plant and insect species (Shinohara et al., 2019).

We organized flood hazard in the study area (**Figure 1A**) by assembling actual flooded areas in 1980 to 2015 provided by the Fukui Prefectural government and the flood hazard map obtained from the National Land Numerical Information download service of the Japanese (Ministry of Land, Infrastructure, Transport and Tourism [MLIT], 2012).

The land-use of urban, agriculture (paddy fields), forests and others in the studied area were identified based on aerial photographs (map data: Google) with enough spatial resolution, and the land-use data (polygons) were digitized in the geographical information system (Arc-GIS, Esri).

Biodiversity surveys

Biodiversity in our study refers to richness or abundance of the taxa that frequently occur in the studied rural landscape consisting of paddy fields and the associated semi-natural grasslands, including plant, butterfly, odonate and frog species. We established a transect (2 m × 30 m) on the levees (semi-natural grasslands) of paddy fields at each sampling site. Surveys of the four taxa were conducted three times on each transect in September 2015, May 2016, and September 2016 when the studied organisms were active and relatively easily observed. For subsequent statistical analyses, we pooled all the data from three surveys. The methodologies of field surveys are as follows:

Plant survey

We established and surveyed three plots (each plot size is 0.5 m × 0.5 m), located at regular intervals (∼10m) along each transect. In total, there were 216 plots in 72 transects. We recorded all vascular plant species in each plot. For statistical

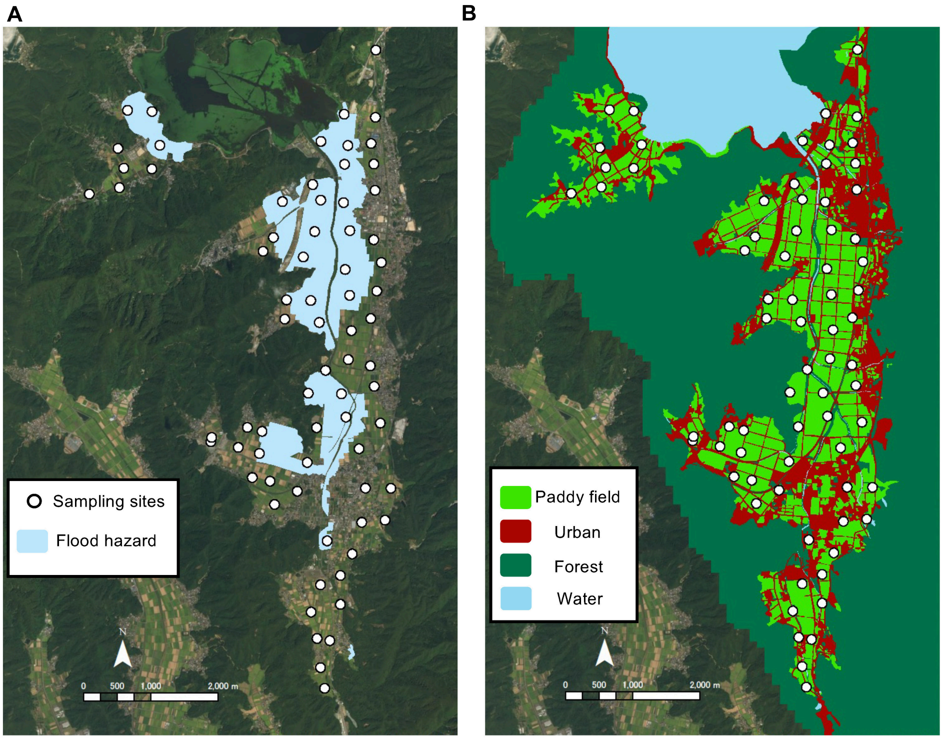

FIGURE 1
Study area in Wakasa town, Fukui Prefecture, Japan. Panel **(A)** shows the sampling sites (white circles) and flood hazard (light blue areas). Panel **(B)** shows the land-use in and around the study area.

analyses, we pooled the data from the three plots located in the same transect.

Butterfly, odonate and amphibian survey

Butterfly and odonate species were identified and the number of two amphibian species (Daruma pond frog (*Pelophylax porosus*) and Japanese wrinkled frog (*Glandirana rugosa*) were counted by a skilled surveyor at the same time with visual observation for 15 min. between 9 a.m. and 4 p.m. within each transect at a height of less than 3 m under warm, sunny conditions as in the previous studies (Pollard and Yates, 1993; Uchida and Ushimaru, 2014). This protocol was decided according to the preliminary survey and identification of species, and we could detect the organisms well enough to see the differences among transects (**Figure 2**). We paid careful attention to avoid double counting of individual organisms. We focused on the two dominant amphibian species as other amphibian species were observed only scarcely.

Analysis of the relationship among flood hazard, land use and biodiversity

To examine how flood hazard and land-use patterns influences various species inhabiting the study area, we constructed a generalized linear mixed model (GLMM) with a poison distribution. We used plant species richness, butterfly species richness, odonate species richness, and abundance of Daruma pond frog and Japanese wrinkled frog as response variables and created a model for each taxon. To analyze the relationships between biodiversity, flood hazard and landscape factors, we adopted whether or not a sampling site was in the flood hazard, and landscape factors in the area of a circle-shaped buffer created around each sampling site, as potential explanatory variables. The landscape factors consist of a combination of three types of land-use (urban, paddy fields and forests; **Figure 1B**) and whether or not the land-use is located in the flood hazard, counting six variables of landscape factors in total. Thus, each landscape variable was an area (m^2) of a type of landscape, so that the coefficients of landscape factors in the same model can be compared to indicate the magnitude of the influence. We also evaluated a degree of farmland consolidation (three levels) at each sampling site and included it as a random effect in the model. Three levels of farmland consolidation were zero, one time, and two times of the consolidation that have been implemented before our study, whose information was provided by the Wakasa town.

Then, we constructed preliminary models with respect to each of different buffer sizes whose radiuses range from 100 m to 1,000 m with an interval of 100 m to identify the best buffer

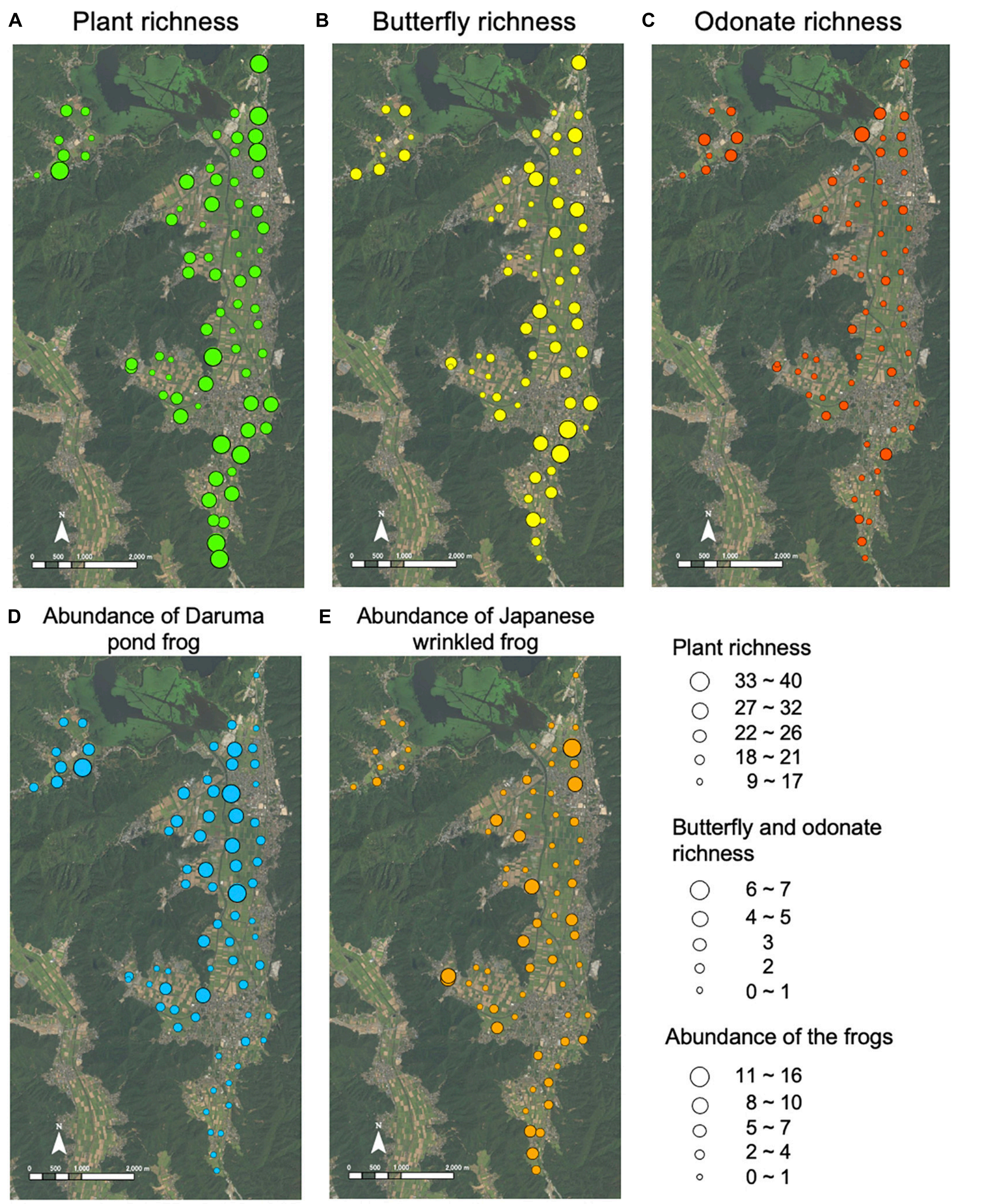

FIGURE 2
Distribution of species richness of plant **(A)**, butterfly **(B)**, and odonate **(C)**, and abundance of Daruma pond frog **(D)** and Japanese wrinkled frog **(E)** in the study area of Wakasa town, Fukui Prefecture, Japan.

size for the landscape factors in the model. In this preliminary analysis, we constructed models using all explanatory variables with a different buffer size, and then selected a model with the lowest AIC (see **Supplementary Figure 1**) to identify the best buffer size for each taxon. After that, we constructed a GLMM for each taxon using the identified best buffer size for the landscape factors. The explanatory variables are six landscape factors (urban, paddy fields and forests × with or without flood hazard) and whether or not the sampling site is in the flood hazard. In total, we constructed 128 models using these explanatory variables with different combinations for each taxon. Models with ΔAIC $<$ 2 were averaged for each taxon using "full-model averaging" as an averaging method according to Lukacs et al. (2010) and Symonds and Moussalli (2011).

In addition, we checked if the residuals of GLMMs showed spatial autocorrelation and found that strong spatial correlation was not evident overall for the taxa examined (**Supplementary Figure 2**).

Scenario analysis

As a measure of Eco-DRR, replacement of urban land-use to agricultural land-use (paddy fields) was considered in our scenario, as it can reduce the exposure of properties to flood hazard and thus lower the potential flood damages, as well as influencing biodiversity due to the change in land-use patterns. We conducted a scenario analysis based on the results of the GLMM models for each taxon that allowed us to project the change of richness or abundance of the taxa we studied. Due to our sampling and statistical designs, we projected only the local change of richness or abundance, but not the regional diversity or the population size covering the whole study area. In this study, we considered two scenarios. The conversion scenario is gradually replacing urban land-use with flood hazard to agricultural land-use to reduce the exposure and damage from flood. In this case, as some residences are converted, it would not be feasible when the human population is increasing. However, like most municipalities in Japan, this town has been experiencing population decline for the last three decades and the trend is projected to continue in the future (Wakasa town, 2015). In the swapping scenario, which would be more feasible even if the human population is not in the decreasing trend, we gradually swapped urban land-use with flood hazard and agricultural land-use without hazard.

In each scenario, we replaced the land-use in a computer simulation described below and projected the richness or abundance of different taxa and the reduction of potential flood damage as an economic benefit. The simulation procedures of the conversion scenario are as follows:

(1). Choose a polygon of urban land-use with flood hazard randomly and then replace an area of d m^2 in the polygon with the same area of agricultural land-use (paddy fields). If the polygon of urban land-use with flood hazard is smaller than d m^2, replace all the urban land-use of the polygon with agricultural land-use.

(2). Repeat the procedure 1 until the total replaced area reaches a targeted replacement rate ranged from 10 to 100%.

(3). Project the richness or abundance of each taxon using the newly gained land-use map and the specific GLMM model.

Here, we set the parameter d to 100 m^2. We tested other parameter values (10, 50, 300 m^2), and confirmed that the results were not different significantly. For the swapping scenario, in the procedure 1, we swapped urban land-use with flood hazard and agricultural land-use without hazard, both of which were randomly selected in the study area. The simulation repeated 100 times at each replacement rate that ranged from 10 to 100% of the total area of urban land-use exposed to flood hazard, by an increment of 10%.

We also calculated the difference in potential flood damage (monetary loss) between before and after the land-use change. The potential flood damage of urban and agriculture land-use was assumed if it was exposed to flood hazard, and the potential monetary loss was calculated based on the actual damages happened in 2013 in the studied municipality. The unit average of the actual damage per 1 m^2 for each land-use was obtained from the data of the survey on flood damages collected by the Ministry of Land, Infrastructure, Transport and Tourism [MLIT] (2019). After the simulation of the land-use replacement, we calculated the expected total damage in the study area by multiplying the exposed area of urban and agricultural land-use to flood hazard and the unit average of the damages for respective land-use (8162 JPY m^{-2} for urban and 29 JPY m^{-2} for agricultural land-use). It should be noted that actual flood damage may be different from our assumption of potential damage depending on the magnitude of actual flood.

Results

In total, we found 242 plant species, 12 odonate species, 22 butterfly species, and 7 amphibian species in our sampling sites. We also observed the total of 253 and 163 individuals of Daruma pond frog and Japanese wrinkled frog, respectively. A list of all species observed in this study was given in **Supplementary Table 1**. The richness of plant, butterfly and odonate and abundance of the two frogs showed marked variations in distribution in the study area (**Figure 2**). The plant richness showed less variations among sampling sites, but still tended to be high at an edge of the forest, where up to forty species were found in the surveyed 0.75 m^2 (**Figure 2A**, coefficient of variation (C.V) = 0.279). Among insects, the butterfly richness spread all over the study area with the maximum of 7 species observed during the survey (**Figure 2B**, C.V. = 0.575), while the odonate richness was relatively high around the lake (Lake Mikata) with the maximum of 4 species (**Figure 2C**, C.V. = 0.869). Daruma pond frogs (*Pelophylax porosus*) seemed to be abundant in the areas with flood hazard with the maximum of 16 individuals observed during the survey (**Figure 2D**, C.V. = 1.016), while Japanese wrinkled frogs (*Glandirana rugosa*) did not show such a trend and up to 11 individuals were found (**Figure 2E**, C.V. = 1.375).

The best buffer sizes of the models explaining the distributions were 700 m for plant richness, 500 m for butterfly richness, 300 m for Daruma pond frog abundance, and 400 m for Japanese wrinkled frog abundance (**Supplementary Figure 1**). For odonate richness, 1,000 m was the best buffer size in our analysis, although the AIC of the model might be lower with a larger buffer size (**Supplementary Figure 1**). However, if we use a buffer size larger than 1,000 m, overlapping among buffers becomes too large to lead a meaningful conclusion of the analysis.

The results of the GLMM models showed no significant effect of exposure to flood hazard (whether or not a sampling site was in flood hazard) and several significant effects of landscape factors in explaining the distribution of the studied taxa (**Table 1**). We found that there was a certain degree of plant richness independent of the environmental parameters we used because the intercept was relatively high (**Table 1**). Agricultural and urban land-use without flood hazard and forest with flood hazard positively affected the plant richness, although the coefficient was much larger for the forest with flood hazard (**Table 1**). For the butterfly richness, urban land-use with and without flood hazard had a positive effect with the much higher coefficient for that with flood hazard, whereas forest with flood hazard had a negative one (**Table 1**). Forest with flood hazard also negatively affected odonate richness, but effects of other landscape factors were not significant (**Table 1**). For Daruma pond frog, agriculture land-use affected Daruma pond frog positively irrespective of the association with flood hazard, and forest with hazard also showed a positive effect with a higher coefficient compared to those of agriculture land-use (**Table 1**). On the contrary, forest without flood hazard showed a positive effect on Japanese wrinkled frog in addition to urban land-use without hazard (**Table 1**). As odonate richness was not significantly influenced by the landscape factors associated with the scenario analysis (urban and agricultural land-use), our scenario of land-use replacement would not lead to a meaningful conclusion for this taxon, so that we excluded odonate richness from the subsequent scenario analysis.

Results of the scenario analyses showed no qualitative difference on the species richness or abundance between the conversion and swapping scenarios (**Figure 3**). For plant species richness and abundance of the two frogs, slightly positive effects of replacing land-use were detected (**Figures 3A,C,D**), although the slightly negative effect was shown for butterfly species richness (**Figure 3B**). For the potential flood damage, we found that the land-use replacement was attributed to reducing the potential damage by 5.19 billion yen (ca. 46 million US$) if we replace urban land-use by agricultural one for all areas with flood hazard (**Figure 3E**). Although the overall effects of land-use replacement on species richness and abundance were not remarkable, there were large variations in the effects depending on the location in the study area (**Figures 4**, **5**). For the conversion scenario, the effects of land-use replacement were relatively large in the northern locations, especially in those with flood hazard, for plant and butterfly richness and abundance of Daruma pond frog (**Figure 4**), although abundance of Japanese wrinkled frog did not show such a spatial pattern. After the conversion, plant richness and Daruma pond frog abundance were expected to increase by up to 3% and 25%, respectively, although butterfly richness was expected to decrease by until −67%. The similar spatial patterns were observed for the swapping scenario (**Figure 5**), although for butterfly richness and abundance of Daruma pond frog, both of positive and negative effects were observed (**Figures 5B,D**), which was not observed in the conversion scenario. After the swapping, plant richness and Daruma pond frog abundance were expected to increase by up to 16 and 24%, respectively, although butterfly richness was expected to decrease by until −68%.

Discussion

Our results demonstrated that local biodiversity including species richness and abundance was shaped interactively by land-use patterns and flood hazard in the agricultural landscape dominated by paddy fields. This is in line with ecological literature that has repeatedly shown the importance of both disturbance and land-use in shaping ecological communities (e.g., Townsent et al., 1997; Pollock et al., 1998; Uchida and Ushimaru, 2014; Shinohara et al., 2019). Also, the results have significant implications for multiple functions of Eco-DRR as seen in our scenario analysis. The land-use replacement to reduce exposure of urban land-use to flood hazard affected positively different taxa we studied, as we hypothesized. In the east lakeside areas associated with flood hazard, the plant richness and Daruma pond frog abundance benefited from the land-use replacement than in the other areas (**Figures 4**, **5**), indicating spatial heterogeneity of the effect. Thus, the lakeside areas have potentially high priorities, in which reduction of exposure of urban land-use to flood hazard as a measure of Eco-DRR is expected to result in both disaster risk reduction and biodiversity conservation. The plant richness was positively influenced by the land-use replacement (**Figure 3A**), because the plant richness was positively correlated with urban land-use without flood hazard and forests with flood hazard (**Table 1**). Also, creating paddy fields and associated semi-natural grasslands near forests would produce positive effects on plant richness on the grasslands, as in previous studies that showed the importance of a mosaic landscape for species such as birds (Amano et al., 2008) and spiders (Miyashita et al., 2012). As for the two frog species, the land-use replacement was expected to benefit Daruma pond frog more than Japanese wrinkled frog (**Figures 3C,D**). This should be due to the difference in habitat preference between the two species. Daruma pond frogs occurred around flood hazard areas (**Figure 2D**), whereas Japanese wrinkled frogs did not show such a trend (**Figure 2E**). The results suggest that species using habitats more related to flood hazard such as wetlands are more likely to benefit from the Eco-DRR measure of reducing exposure of urban land-use to flood hazard.

Although the land-use replacement showed the positive effects on some taxa, it negatively affected butterfly species. Most of the butterfly species observed in the study area were common species (**Supplementary Table 1**), and they might be

10.3389/fevo.2022.699201

TABLE 1 Results of the GLMM models explaining the distribution of the taxa studied.

	Intercept	Exposure to flood hazard	Landscape factors						Buffer size (m)	R-squared
			Agriculture		Urban		Forest			
			With flood hazard	Without flood hazard	With flood hazard	Without flood hazard	With flood hazard	Without flood hazard		
Plant richness	2.457*** (0.171)	−0.006 (0.042)	0.178 (0.295)	1.139*** (0.224)	0.017 (0.337)	1.354*** (0.408)	9.844* (4.410)	−0.024 (0.151)	700	0.251
Butterfly richness	0.411 (0.249)	0.012 (0.100)	0.030 (0.219)	−0.115 (0.375)	23.912* (10.583)	2.274** (0.770)	−5.162* (2.361)	0.031 (0.229)	500	0.270
Odonate richness	0.782 (3.166)	−0.007 (0.097)	9.460 (10.143)	−0.177 (0.607)	56.732 (99.799)	−0.527 (1.438)	−2.764* (1.299)	−2.885 (2.767)	1000	0.196
Daruma pond frog abundance	−0.929* (0.399)	−0.009 (0.096)	3.103*** (0.479)	2.966*** (0.557)	−0.215 (2.123)	0.105 (0.404)	6.607*** (1.006)	−0.066 (0.361)	300	0.478
Japanese wrinkled frog abundance	−1.112* (0.523)	0.064 (0.198)	0.165 (0.477)	−0.071 (0.313)	−2.001 (12.155)	4.417*** (0.901)	−8.733 (11.878)	3.092*** (0.714)	400	0.145

Explaining variables included whether or not a sampling site was in flood hazard (exposure to flood hazard) and landscape factors in a circle buffer created around each sampling site. ***$p < 0.001$, **$p < 0.01$, *$p < 0.05$. Values represent estimated coefficients and values in parentheses refer to standard errors.

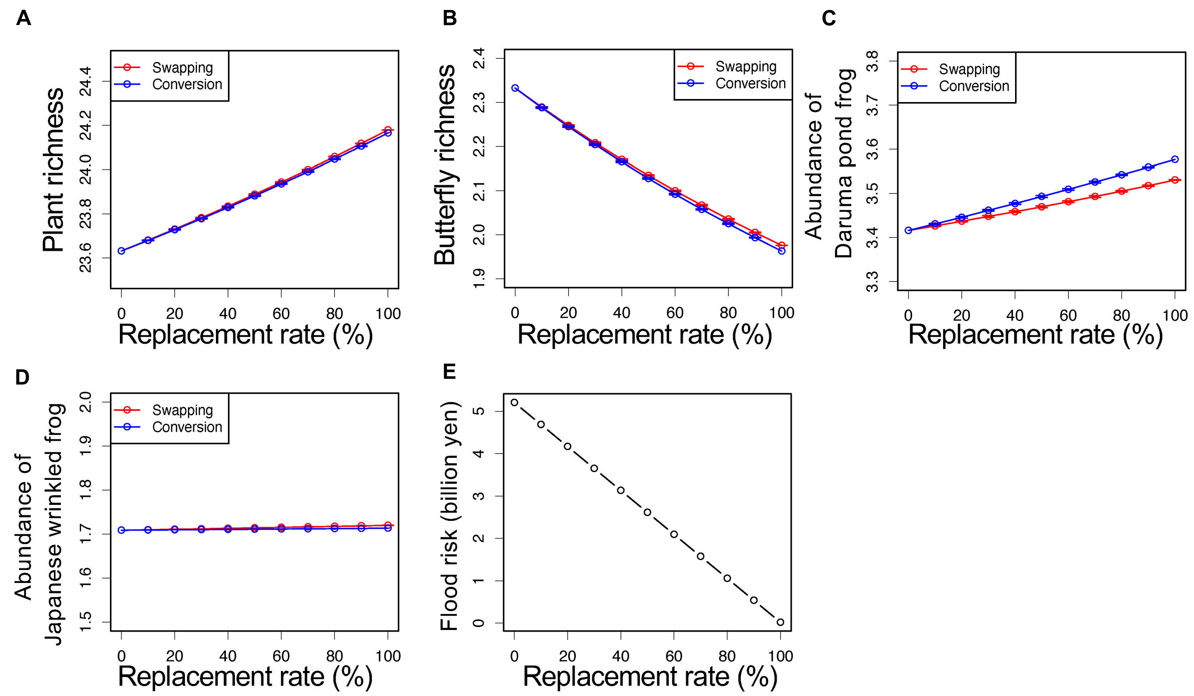

FIGURE 3
Results of the two scenario analyses of land-use replacement on species richness (plant **(A)** and butterfly **(B)**) and abundances of frogs (Daruma pond frog **(C)** and Japanese wrinkled frog **(D)**) and the potential flood damage **(E)**. Red lines and blue lines represent the swapping and the conversion scenarios, respectively, with the mean and standard deviation for repeated simulations at each replacement rate. Standard deviation was generally too small to be visible in the figure. Note that the potential flood damage was not different between the two scenarios.

dependent on artificial vegetation grown in domestic gardens. In an urban landscape, sunlight and floral abundance are important factors in determining the diversity of pollinator communities including butterflies (Matteson and Langellotto, 2010) and domestic gardens have an important role in supporting biodiversity (Cameron et al., 2012). In the study area of a rural landscape, there was no tall building, but domestic gardens still might have played an important role to support pollinator species.

Whether the sampling site itself was exposed to flood hazard was not important in shaping local biodiversity in our study (**Table 1**). This may be related to the fact that our study sites were paddy fields and associated semi-natural grasslands, and that paddy fields experience artificial floods (or irrigation) from spring to summer seasons for rice growing, which is similar timing to that of natural floods caused by the monsoon. Indeed, paddy fields and associated semi-natural grassland provide wet to moist environments, so that they can be an alternative habitat for organisms living in natural wetlands and floodplains (e.g., Kiritani, 2010; Natuhara, 2013). Thus, natural flood itself might have less influence on local biodiversity in the study sites, compared to the landscape factors we considered.

Our results also showed the potential financial benefits of 5.19 billion yen (**Figure 3E**) if we can replace all urban land-use in the flood hazard areas to agricultural one. This was probably an overestimation because we assumed that all houses in the flood hazard areas were lost and fully damaged if floods actually occurred. The assumption resulted in a linear decline of potential flood damage as the replacement rate increases (**Figure 3E**). In reality, the flood damage can be variable depending on the local situations including drainage distance, elevation, and flow accumulation (Kousky and Walls, 2014; Kazakis et al., 2015). Recently, technologies of mapping flood hazard have advanced to be able to show very details of flood hazard (e.g., Taki et al., 2013; Motevalli and Vafakhah, 2016; Razavi-Termeh et al., 2018). If we can combine the details of flood hazard obtained by the state-of-the-art techniques with our scenario analysis, we should be able to provide better information about, for example, high priority areas that would be more relevant to the land-use policy and management. Also, our analysis did not account for human damage that is also important for disaster risk reduction. Avoiding exposure to flood hazard by the land-use replacement is expected to reduce human damage as well, although this remains to be considered in the future research.

There was no remarkable distinction in the projected species richness and abundance between the conversion and swapping scenarios (**Figure 3**). As the difference between the two scenarios was the land-use outside the flood hazard areas, the results suggest that the land-use within the flood

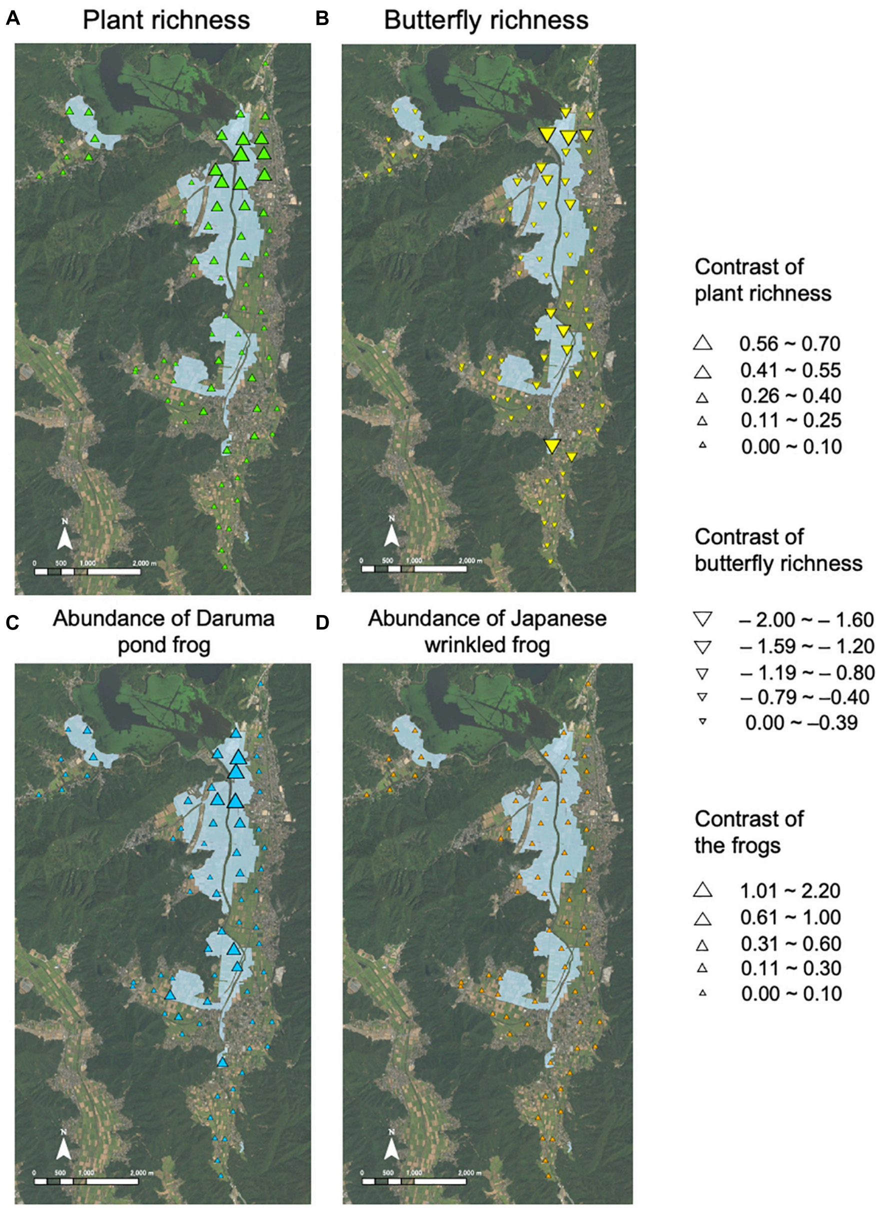

FIGURE 4
Contrast of plant richness **(A)**, butterfly richness **(B)**, and abundances of Daruma pond frog **(C)** and Japanese wrinkled frog **(D)** between before and after the land-use replacement in the conversion scenario. Light blue areas show flood hazard.

hazard areas was more important in shaping local biodiversity than that outside the flood hazard areas. The flood hazard areas were originally natural wetlands and floodplains before paddy fields were constructed in ancient times, and paddy fields provide alternative habitats to species closely associated with natural wetlands (Natuhara, 2013; Katayama et al., 2015). Thus, land-use change from urban to agriculture (paddy fields) within the flood hazard areas might have more influence on local biodiversity than the land-use change outside the flood hazard areas. However, in the conversion scenario, habitat area itself increases in addition to the landscape changes, and this should have large effects on local biodiversity as well, although our analysis did not allow us to examine such effects. In addition, implementing the swapping scenario in reality should be associated with more costs for constructing residence areas than implementing the conversion scenario without such a new construction, although the expected reduction of flood damage is the same between the two scenarios and the conversion scenario assumes the decline of human population. These socio-economic conditions in the local community were also not considered in our scenario analysis, although scenario analysis can be more flexible to include complexities and trade-offs

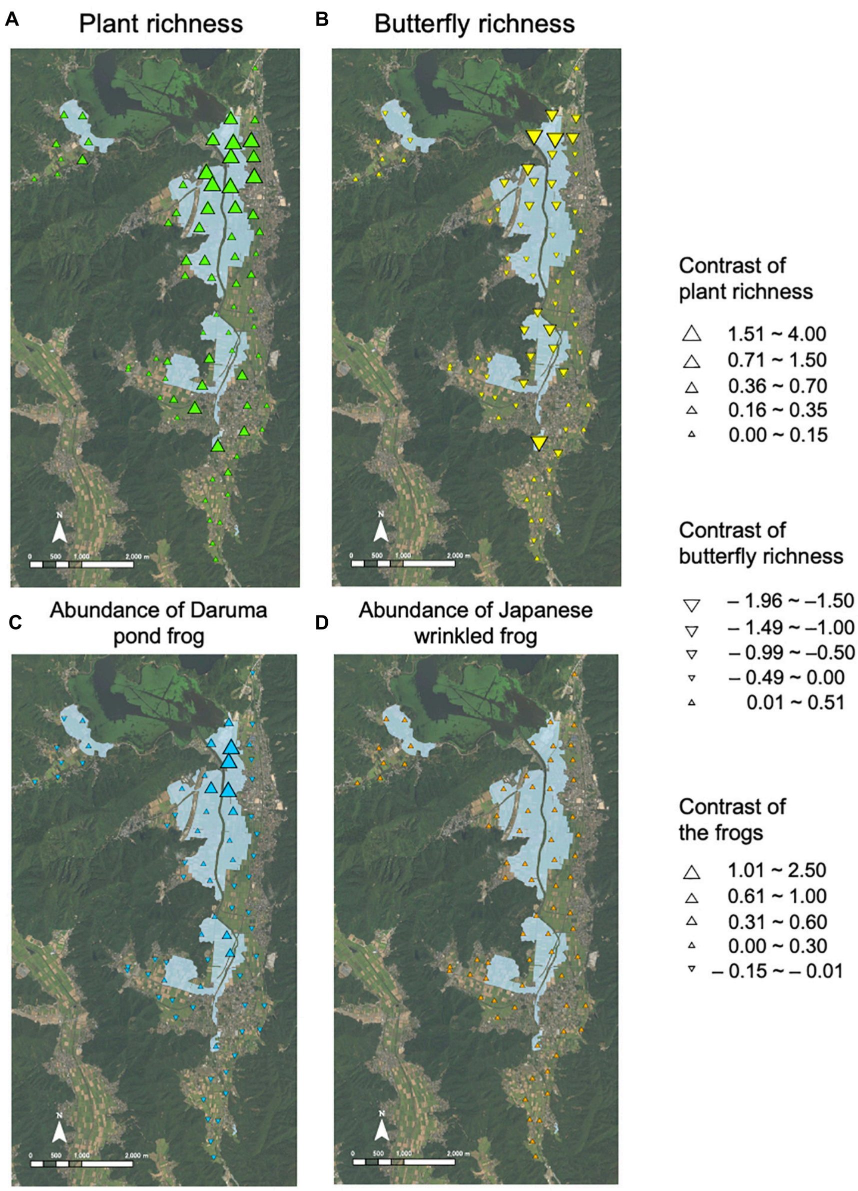

FIGURE 5
Contrast of plant richness **(A)**, butterfly richness **(B)**, and abundances of Daruma pond frog **(C)** and Japanese wrinkled frog **(D)** between before and after the land-use replacement in the swapping scenario. Light blue areas show flood hazard.

inherent in decision making processes (Sayers et al., 2002; Casal-Campos et al., 2015).

Our scenario analysis considered the replacement from urban to agricultural land-use to reduce exposure to flood hazard. However, it may be more effective for biodiversity conservation to replace to wetlands instead of agriculture land-use. Wetlands not only support high productivity and biodiversity but also provide important ecosystem services including flood regulation (Gibbs, 2000; Woodward and Wui, 2001; Zedler and Kercher, 2005; Everard et al., 2009). Thus, in the context of Eco-DRR, replacing urban land-use to wetlands would be more effective. Nevertheless, the scenario analysis with wetlands was not possible due to the lack of the data sets for wetlands that were very scarce in the study area. It would be an effective approach to restore wetlands even in a small area and then obtain more information of such wetlands to construct an alternative scenario with the land-use replacement to wetlands.

Reconstructing wetlands can be an alternative effective approach that provides multiple ecosystem services (Everard et al., 2012).

We conducted the scenario analysis to quantify the effects on local biodiversity and flood damage reduction if the land-use replacement (either conversion or swapping) to reduce exposure of urban land-use to flood hazard is opted for an actual measure of Eco-DRR. The results suggest that the Eco-DRR approach can dramatically reduce the potential flood risk and provide overall positive effects on local biodiversity, although the effects were different depending on species and spatial location, as we hypothesized. The importance of agricultural land-use in biodiversity conservation would be relevant not only at a local scale, as shown in this study, but also at a broader scale, for example, in conserving an endangered migrating bird like the Oriental White Stork (*Ciconia boyciana*) that uses the paddy fields as their habitat (Yamada et al., 2019; Tawa and Sagawa, 2021). Thus, land-use planning at a local scale is critically important for both reducing flood risk and conserving local biodiversity, which the implementation of Eco-DRR aims for. Also, it is highly relevant to the global goals and initiatives such as the Sustainable Development Goals, the UN Decade on Ecosystem Restoration, the Sendai Framework for Disaster Risk Reduction, etc. The scenario analysis as we conducted here should provide useful information for quantifying and visualizing the potential outcomes of such land-use planning to support decision-making at a local scale.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Ethics statement

Ethical review and approval was not required for the animal study because we conducted field observations of amphibians without collecting them.

Author contributions

MK, KU, NS, and TY contributed to conception and design of the study. KU and NS conducted field surveys. MK performed the scenario analysis. All authors wrote the manuscript and approved the submission.

Funding

This study was supported by the Environment Research and Technology Development Fund (4-1505) of Environmental Restoration and Conservation Agency, the Research Institute for Humanity and Nature (RIHN: a constituent member of NIHU) Project No. 14200103, and the Grant-in-Aid for JSPS Fellows (19J00864) (to MK).

Acknowledgments

We thank Satoru Itagawa, Yumi Yamada, Mifuyu Ogawa, Jun Ishii, Takuya Yoshida, Masaki Hirano, Keidai Iiyama, and Chinatsu Zakouji for their assistance. We also thank the reviewers and TM for their constructive comments and suggestions.

Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

The handling editor TM declared a shared affiliation with the authors KU and TY at the time of the review.

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Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fevo.2022.699201/full#supplementary-material

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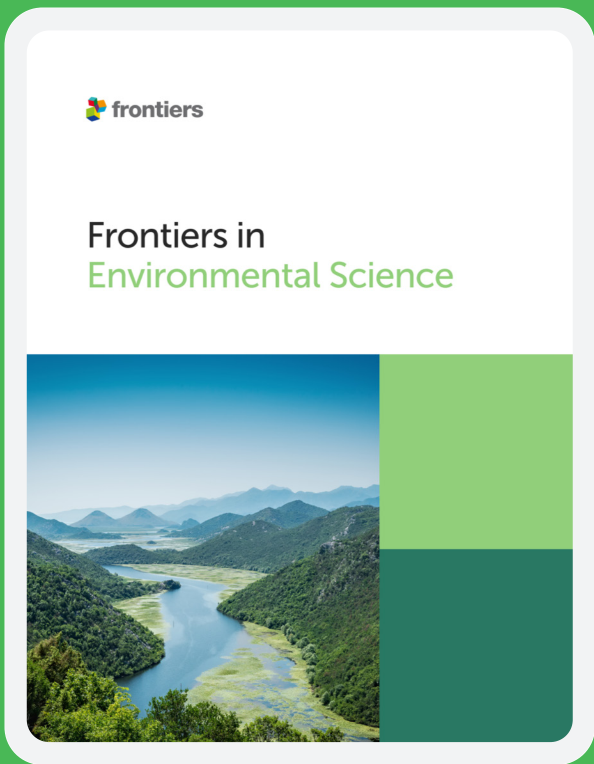

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