UNIVERSIDADE DE LISBOA
FACULDADE DE CIÊNCIAS
DEPARTAMENTO DE BIOLOGIA ANIMAL

# Conserving biodiversity and enhancing food security: understanding small mammal diversity patterns in mosaic landscapes of Guinea-Bissau (West Africa)

Raquel Nogueira de Oliveira

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Dissertação orientada por:
Doutora Ana Filipa Palmeirim
Doutora Ana Rainho

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“Our task must be to free ourselves [...] by widening our circle of compassion to embrace all living creatures and the whole of nature in its beauty.”

**— Albert Einstein**

## Resumo

A necessidade de atender às crescentes exigências da população humana enquanto se garante a conservação da biodiversidade constitui um verdadeiro desafio. Os esforços atuais para aumentar a produção a fim de satisfazer as necessidades humanas baseiam-se maioritariamente na expansão agrícola com a adoção de práticas de monocultura. No entanto, estas práticas têm diversos impactos ecológicos e não constituem uma fonte eficiente de produção a longo prazo, pelo que é necessário arranjar soluções mais ecológicas. Isto é particularmente relevante em países tropicais em desenvolvimento, onde a insegurança alimentar coexiste com elevados níveis de biodiversidade, e ainda mais urgente em paisagens afrotropicais, dado que a maioria dos estudos relativos a paisagens agrícolas foi realizado nos neotrópicos. Na Guiné-Bissau, florestas nativas e arrozais tradicionais estão a ser convertidos em monoculturas de caju (*Anacardium occidentale*) a taxas sem precedentes. Acredita-se que esta alteração afete negativamente a vida selvagem, diminuindo a riqueza de espécies e favorecendo espécies generalistas. Somando tudo isto à instabilidade política verificada na África Ocidental, há uma necessidade urgente de dar prioridade à gestão sustentável para alcançar a autossuficiência e a soberania alimentar. Alguns exemplos de soluções baseadas na natureza incluem, por exemplo, a integração de paisagens em mosaico que contenham manchas de habitat nativo entre as paisagens humanizadas e a gestão ecológica de pragas. No entanto, um pré-requisito para implementá-las de forma eficiente é identificar as espécies presentes e compreender como diferentes grupos usam estas paisagens.

Este estudo pretende investigar os padrões de diversidade de pequenos mamíferos em paisagem rurais da Guiné-Bissau em três tipos de habitat: florestas, bolanhas (nome local para arrozais) e plantações de caju, a diferentes escalas. Foram amostrados 15 locais, em torno de 5 aldeias. A uma escala local, ou seja, considerando apenas o tipo de habitat em que a amostragem decorreu, pretendemos comparar a composição das espécies, a riqueza e o índice de diversidade de Shannon. À escala da paisagem, ou seja, considerando a proporção de cada habitat dentro de um *buffer* específico, pretendemos compreender como é que os diferentes habitats influenciam a riqueza e a abundância deste grupo. Espera-se que a riqueza seja maior em florestas devido à maior complexidade do habitat e que a abundância seja maior em bolanhas dada a elevada produtividade do arrozal, favorecendo as taxas de reprodução de algumas espécies. Espera-se também que as plantações de caju tenham um efeito negativo tanto na riqueza como na abundância das espécies, uma consequência comum de sistemas de monocultura. Consequentemente, esperamos que a composição das espécies seja diferente entre os três tipos de habitats.

Os pequenos mamíferos foram amostrados usando armadilhas Sherman e foram identificados ao nível taxonómico mais baixo possível através de métodos moleculares. Em cada local de amostragem foi realizado um transeto que consistiu em 10 armadilhas Sherman espaçadas por intervalos de 10 metros. A amostragem decorreu em três épocas de amostragem ao longo do ano, sendo que em cada uma as armadilhas ficaram ativas durante oito noites consecutivas.

Foram capturados 230 indivíduos pertencentes a 11 espécies. Numa escala local, a composição de espécies de pequenos mamíferos diferiu entre os habitats, sendo a comunidade presente nas bolanhas particularmente distinta. É também de salientar que as espécies encontradas nas plantações de caju foram também encontradas nos restantes tipos de habitat, apoiando a noção de que as plantações de monocultura tendem a favorecer espécies generalistas. No entanto, a riqueza de espécies e o índice de diversidade de Shannon não diferiram entre os três habitats. Estes resultados salientam a necessidade de ter em conta outros parâmetros

além da escala local, nomeadamente a proximidade dos transetos de amostragem a outros tipos de habitat, ou a proporção de cada habitat e quais os habitats que o rodeiam numa paisagem mais ampla. Efetivamente, os nossos resultados a uma escala mais ampla estão de acordo com esta afirmação, tendo sido detetado o efeito de dois tipos de habitat.

À escala da paisagem, tanto a riqueza como a abundância foram mais elevadas em paisagens com maior proporção de bolanhas, destacando a importância deste habitat para os pequenos mamíferos. Este resultado não suporta a hipótese de que a riqueza de espécies seria maior em florestas. A principal razão para esta observação está provavelmente ligada à elevada disponibilidade de água nas bolanhas. Ao contrário de outros habitats, as bolanhas sofrem inundações sazonais, proporcionando uma fonte consistente de água que pode ser escassa em outras áreas. Esta disponibilidade de água nas bolanhas promove a abundância de insetos, beneficiando pequenos mamíferos insetívoros, como os musaranhos, enquanto o arroz serve de alimento para espécies granívoras, ambos favorecendo também as espécies omnívoras. Deste modo, as bolanhas tornam-se ambientes bastante ricos durante uma parte significativa do ano, promovendo a diversidade e constituindo ambientes ideais para a reprodução. É também importante reconhecer que as florestas incluídas neste estudo não são pristinas e são ativamente utilizadas pelas comunidades locais, apresentando sinais de degradação, o que pode também ter contribuído para os padrões observados. Por outro lado, a abundância foi menor em proporções mais elevadas de plantações de caju, refletindo a potencial menor disponibilidade de recursos neste habitat, tanto em termos estruturais como tróficos.

No geral, os nossos resultados apoiam a ideia de que as paisagens em mosaico são benéficas para os pequenos mamíferos, particularmente aquelas com presença de floresta, maior cobertura de bolanha e proporções reduzidas de plantações de caju. Este estudo serve como conhecimento base para a compreensão dos padrões de diversidade de pequenos mamíferos numa paisagem em mosaico, para que seja possível desenvolver estratégias e políticas de gestão personalizadas destinadas a aumentar a produtividade das culturas e a segurança alimentar, visando simultaneamente abordar os desafios relacionados com as pragas e preservar a biodiversidade. Esta abordagem pode promover uma melhor coexistência e ajudar a travar a tendência devastadora de adoção de práticas ecologicamente prejudiciais. O aumento da heterogeneidade da paisagem intrínseco a esta abordagem é valioso do ponto de vista ecológico e também uma salvaguarda para as comunidades humanas locais não dependerem apenas das monoculturas como fonte de rendimento, uma vez que são mais vulneráveis a pragas e doenças. De qualquer forma, manter manchas de florestas nativas ou restaurar as características do habitat natural degradado dentro destas paisagens de forma a melhorar a qualidade da composição da paisagem é crucial para conservar a biodiversidade de variados grupos e reduzir os impactos negativos da alteração do uso do solo.

É essencial reconhecer que a área de estudo é altamente heterogénea, caracterizada por uma paisagem composta por diferentes tipos de habitat, como florestas, bolanhas e plantações de caju, próximos uns dos outros. Como resultado, os “efeitos de borda”, que se referem às influências que as zonas de transição entre diferentes tipos de habitat têm nas comunidades animais, são relativamente grandes e difundidos em toda a paisagem. Em contraste, as áreas dentro de cada habitat onde as condições ambientais típicas permanecem relativamente não perturbadas por estas zonas de transição, o “núcleo” de cada habitat, são limitadas em extensão e influência, resultando numa diminuição da sua importância ecológica. Isto pode resultar em comunidades mais homogéneas e potencialmente reduzir a diversidade beta global, especialmente em animais maiores que podem não conseguir satisfazer os seus requisitos numa paisagem composta por

manchas de habitat mais pequenas. Estudos futuros poderiam aprofundar a análise desses efeitos de borda e o seu potencial impacto nos padrões de diversidade de espécies em paisagens em mosaico e continuar a estudar a influência de diferentes paisagens em mosaico em diferentes grupos.

Adicionalmente, acreditamos que a conservação só pode ser feita com a população local. Os esforços de conservação devem envolver a participação e colaboração das comunidades locais com interesse e influência diretos nas questões ambientais. A conservação com pessoas implica também que as dimensões sociais, culturais e económicas da conservação sejam tidas em conta, incluindo a segurança alimentar e a provisão de recursos para as comunidades locais. Uma forma de garantir a segurança alimentar e melhorar a produção agrícola, conservando simultaneamente a biodiversidade, pode ser alcançada através de estratégias de intensificação ecológica. Os resultados deste estudo contribuem também para a fundação de uma base robusta de conhecimento para as espécies presentes na área de estudo, estabelecendo as bases para a aplicação de estratégias de gestão ecológicas de pestes de roedores.

Em suma, este estudo fornece informações sobre a diversidade de pequenos mamíferos em paisagens em mosaico da Guiné-Bissau, salientando a importância das bolanhas. Estes resultados fortalecem o potencial papel das paisagens heterogéneas compostas por manchas agrícolas e habitat nativo em complementar os esforços de conservação e abrem caminho para o desenvolvimento de técnicas de gestão de produção e controlo de pragas eficazes adaptadas a contextos locais específicos, visando satisfazer também as necessidades humanas.

**Palavras-chave:** Agricultura, Alteração do uso da terra, Arrozais, Expansão do caju, Gestão de roedores

# Abstract

The need to meet human population increasing requirements while ensuring biodiversity conservation has challenged current management practices and called for nature-based solutions. This is particularly relevant in the understudied Afrotropical landscapes where food insecurity meets high levels of biodiversity. These nature-based solutions can include the integration of mosaic landscapes containing patches of native habitat within human-modified landscapes and ecologically based pest management. However, a prerequisite to implement these solutions efficiently is to understand how different groups use and persist across human-modified landscapes and identifying the species present. This study investigates small mammal diversity patterns (species richness, Shannon index, abundance and composition) in rural landscapes of Guinea-Bissau at different scales in three habitats: forests, *bolanhas* (local name for rice paddies) and cashew orchards. Small mammals were sampled across 15 sampling sites using Sherman live traps. We captured 230 individuals belonging to 11 species. At a local scale, small mammal composition differed across the habitats, with *bolanhas* supporting distinctive assemblages. However, species richness and Shannon diversity index remained similar between the three habitats. At a landscape scale, both observed species richness and abundance were higher in landscapes with higher proportion of *bolanha*, highlighting the importance of this habitat for small mammals. Conversely, abundance was lower under higher proportions of cashew orchards, reflecting the potentially lower availability of resources in this habitat. Overall, our results support the idea that mosaic landscapes are beneficial for small mammals, particularly those with a reasonable *bolanha* coverage and reduced proportions of cashew orchards. These findings strengthen the potential role of heterogeneous landscapes comprising agricultural crops and native habitat in fostering biodiversity conservation. This study serves as a baseline knowledge to develop tailored management strategies and policies aimed at enhancing crop productivity and food security while concurrently addressing pest-related challenges and preserving biodiversity across tropical modified landscapes.

**Keywords:** Agriculture, cashew expansion, land-use change, rice paddy, rodent-management

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# 1. Introduction

## 1.1. Agricultural expansion and biodiversity decline

The demand for agricultural resources intensifies as the human population continues to grow, and finding a way to sustainably meet these needs is of global concern (Bajželj et al., 2014; Tilman et al., 2011). Efforts to maximise food production to meet this demand drive agricultural expansion with the widespread adoption of monoculture practices replacing native vegetation (Laurance et al. 2013; Monteiro et al., 2017; Pimentel et al., 1992; Tscharntke et al., 2012). While these practices may meet immediate human needs, their long-term efficiency is questionable since they lead to soil nutrient depletion and aggravate the vulnerability to pests and diseases (Matson et al. 1997; Laurance et al. 2013; Tilman et al., 2011). Current agricultural practices have also resulted in a series of adverse ecological consequences, which include fragmentation of forests and other native habitats (Hansen et al., 2020), landscape homogenisation (Jeliazkov et al., 2016), species loss (Hurst et al., 2013; Scales & Mardsen, 2008), shifts in species composition (Newbold et al., 2016) and declines in functional diversity (Flynn et al., 2009; Hurst et al., 2013), ultimately aggravating the ongoing biodiversity crisis (Robertson & Swinton, 2005; Tscharntke et al., 2012).

Although it is widely known that land-use change is a major driver for biodiversity loss, the response of biodiversity varies among geographical regions (Newbold et al., 2016), and the degree to which specific taxa vary between land-use types remains unclear (Foord et al., 2018; Newbold et al., 2014; Reynolds et al., 2018). This is particularly pertinent in developing tropical countries, where food insecurity, high land-use intensity and an increasing human population rate coincide with high levels of biodiversity (Newbold et al., 2020). This is even more relevant in Afrotropical landscapes, given that most research on tropical agricultural landscapes was conducted in the Neotropics (Powers et al., 2011; Stocks et al., 2008). West Africa and, by extension, Guinea-Bissau, exist within a region of notable biodiversity, hosting globally significant forest and savanna patches (Happold & Lock, 2013). In Guinea-Bissau, rice is essential to local people, being the most consumed food item and a significant source of employment and livelihood for most of the population (Medina, 2008). Despite this, native forests and traditional rice fields in the country are being converted into cashew monocultures (*Anacardium occidentale*) at unprecedented rates (Monteiro et al., 2017; Pereira et al., 2022; Temudo & Abrantes, 2014). This alteration is thought to negatively affect wildlife, decreasing species richness while favouring generalist species (Hurst et al., 2013; Mamba et al., 2019; Scales & Mardsen, 2008; Vasconcelos et al., 2015). Cashew was introduced in several tropical countries by the Portuguese, including Guinea-Bissau, where this cash-crop plays an important role in defining land tenure and ownership (Catarino et al., 2015). This country strongly relies on this kind of culture, given that the export of cashew nut shells constitutes 90% of the country’s total exports (FMI, 2015). Despite its small size, Guinea-Bissau is among the top five largest producers of raw cashew nuts (Catarino et al., 2015). While the expansion of cashew production in Guinea-Bissau is substantial, the exact extent remains unquantified. However, global cashew cultivation has grown from 9,482.6 km² to 65,648.2 km² between 1980 and 2021 (FAOSTAT, 2023). Still, global cashew production is expected to increase even more due to the high demand primarily driven by its recognized health advantages and multifaceted use in various market-derived products (Dendena & Corsi, 2014; Oliveira et al., 2020). Given the countries' dependence on fluctuating cashew prices and the political instability and governance challenges of West Africa, there is an urgent need to prioritise sustainable management for achieving self-sufficiency and food sovereignty.

## 1.2. Importance of mosaic landscapes in conservation

Habitat loss and degradation are the main drivers of biodiversity loss (Gibson et al., 2011; Almeida-Maués et al., 2022). Creating Protected Areas (PAs) is the most effective form of conserving biodiversity and preserving ecological integrity (Laurance et al., 2012; Sodhi & Ehrlich, 2010). However, there is little scope for further expansion of PAs, especially in developing countries, due to densely populated areas, competing land uses, political will, and the limited amount of land available (Jenkins & Joppa, 2009; Komanduri et al., 2023; Venter et al., 2014). Moreover, agriculture is necessary for human sustenance, underscoring the need to reconcile biodiversity conservation and food production. A promising approach involves retaining native patches within human-modified landscapes, promoting biodiversity persistence while providing ecosystem services. Heterogeneous landscapes, comprising mosaics of native habitats and agricultural fields, are increasingly gaining conservation attention in the tropics (Arroyo-Rodríguez et al., 2020; Rege et al., 2020; Vogel et al., 2023). As such, enhancing landscape compositional heterogeneity might mitigate the negative impacts of intensive farming and foster diversity across multiple biological groups in agricultural mosaics (Benton et al., 2003; Fahrig et al., 2011; Reynolds et al., 2018). For instance, if managed within a mosaic containing patches of native habitat, agricultural fields can supplement conservation efforts by maximizing heterogeneity and diversity of vegetation, thus providing a variety of nesting sites, food sources, and suitable foraging habitats for various species (Chazdon, 2008; Chazdon et al., 2009; Rege et al., 2020). Indeed, such an approach might help preserving biodiversity integrity while ensuring essential ecosystem services and provision of resources vital for human well-being, ultimately striking a balance between nature conservation and human needs (Arroyo-Rodríguez et al., 2020; Fulgence et al. 2022, Reynolds et al., 2018; Tscharntke et al., 2012). However, an essential prerequisite for achieving this goal is understanding how different biological groups use and persist across human-modified landscapes. Furthermore, analysing diversity metrics at multiple scales allows for a more holistic understanding of the community, given that different ecological processes operate at different scales (Arroyo-Rodríguez et al., 2020; Willis & Whittaker, 2002). For example, microhabitat conditions influence species diversity at a local scale, while factors like habitat connectivity or the surrounding landscape affect species diversity at larger scales. Thus, we should assess both local land use type and landscape composition (see Carvalho et al., 2023; Mérő et al., 2015; William et al., 2023) to design more effective conservation strategies.

## 1.3. Small mammals and their role in the ecosystem

In Africa, non-volant small-bodied mammals typically comprise species from both Rodentia and Eulipotyphla orders. Overall, small mammals provide a variety of ecosystem services, such as seed dispersion (Terborgh et al., 2001), pollination (Goldingay et al., 1991), arthropod predation (Posa et al., 2007), and serve as prey for higher trophic levels (Hughes et al., 1994; Kotler, 1984). On the other hand, small mammals, particularly rodents, are also known to act as crop pests (Abdel-Rahman Ahmed et al., 2008; Makundi et al., 1999; Monadjem et al., 2015), causing substantial production loss (Htwe et al., 2021; Singleton et al., 2005). In Africa, over 25 rodent species have been documented as agricultural pests, causing significant damage and losses to a variety of crops, including rice (Brown et al., 2017; Donga et al., 2022; Makundi et al., 1999; Swanepoel et al., 2017). Considering the diverse roles small mammals play in ecosystems, it is crucial to understand their responses to land-use changes. These mammals comprise a highly diverse group, exhibiting a wide range of habitat preferences, niche specialization, and dietary habits. Indeed, species can occupy various ecological niches, both in terms of habitat type (e.g., forest, savanna and anthropogenic habitats), vertical strata (semi-fossorial, terrestrial, scansorial or arboreal) and diet-wise (e.g., granivorous, insectivorous or herbivorous). This high diversity within the group implies that their responses to changes in land use might be complex and context-dependent. While alterations in land-use can lead to shifts in small mammal assemblage composition, these responses may not always be negative and thus not always leading to a decrease in either species richness or abundance. In fact, some species may thrive in disturbed habitats due to increased resource availability and reduced predation pressure from larger animals that cannot bear such alterations (Brown et al., 2017; Byrom et al., 2014; Hilty et al., 2006; Laurance & Vasconcelos, 2004; Niang et al., 2022). Nevertheless, small mammals respond quickly to ecosystem cues due to their R-strategy traits (Hurst et al., 2013; Singleton et al., 2010). Therefore, examining shifts in small mammal assemblages while contemplating species identity provides valuable insights into the consequences of land-use changes and ecosystem disturbance (Avenant & Cavallini, 2007; Hoffmann & Zeller, 2005).

Contrary to native vegetation patches, vegetation of crop fields is usually less complex and often subject to abrupt changes of composition and structure both in space and time due to management practices (Loggins et al., 2019; Tooker et al., 2020). Crop fields are thus unable to provide the same structural, climatic and trophic resources as native vegetation (Hardwick et al., 2015). In particular, crop vegetation height displays high temporal variation due to plant growth. In the case of rice fields, vegetation is negligible at the time of planting and very thick and dense just before harvesting (Gheler-Costa et al., 2013), while in cashew plantations vegetation changes are due to management activities such as undergrowth clearing (Camprodon & Brotons, 2006). Despite this, crop fields are productive and may provide abundant, even if the short-term, trophic resources (Boesing et al., 2022). For example, in irrigated ecosystems such as rice fields, water and food are transient, posing no constraints on rodent reproduction and creating optimal conditions for their proliferation (Brown et al., 2017; Niang et al., 2022), especially generalist species that are thought to not have particular requirements (Hurst et al., 2013). Several studies have shown an increase in small mammal abundance in agricultural areas, both in general (Byrom et al., 2014; Foord et al., 2018) and for specific species (Mamba et al., 2019).

One way to increase agricultural production while preserving biodiversity is through nature-based solutions in agricultural management (Pender, 1998; Pimentel et al., 1992; Tscharntke et al., 2012). Ecologically-based rodent management (EBRM) represents an integrative strategy for addressing rodent-related pest issues that mitigates crop losses by rodents and the risks of rodent-borne diseases in an environmentally friendly way by comprehensively exploring the diversity, behaviour, ecology, and population dynamics of rodents, as well as considering human perceptions and practices (Makundi et al., 1999; Singleton et al., 2021). Notably, this approach has successfully enhanced rice and maize production systems across Asia, leading to elevated yields and increased farmer income (Singleton et al., 2021). EBRM encompasses a diverse set of management practices that are incorporated into community-driven initiatives tailored to specific local contexts. These practices may encompass a number of habitat adjustments that render environments less favourable for rodent pests (e.g., reducing vegetation cover and bank length, debris removal and sanitation improvements), including biological control through natural predators, and targeted and controlled use of rodenticides to minimise environmental impact (e.g., deploying bait stations or applying rodenticides exclusively in areas with high rodent populations) (Brown et al., 2017; Donga et al., 2022; Singleton et al., 2021). Nevertheless, although EBRM exhibits the potential for versatile application across a wide range of agro-ecosystems, its implementation in Africa has been notably slow and underexplored (but see Makundi et al., 2011; Swanepoel et al., 2017). This delay can be primarily attributed to the scarcity of foundational research, namely concerning the taxonomy and ecology of rodents. Thus, identifying small mammal species in an ecosystem and understanding their diversity patterns is extremely important for developing targeted methods to minimise crop damage and ensure food security while conserving biodiversity. Apart from being included in the understudied African agricultural landscapes previously mentioned, small mammal assemblages in Guinea-Bissau are largely unknown, highlighting the importance of this study.

### 1.4. Objectives and hypothesis

In this study, we aim to examine patterns of small mammal diversity in rural landscapes at multiple scales by comparing species diversity in three habitat types: forests (native vegetation), rice paddies (subsistence crops, hereinafter referred to by their local name *bolanhas*) and cashew orchards (a commercial crop). More specifically, we intend to:

1. Compare species composition, species richness and Shannon diversity index across the three habitats at a local scale, i.e., considering solely the habitat type in which the sampling took place;
2. Understand how the different habitats influence species richness (observed and estimated) and abundance at a landscape scale, i.e., considering the proportion of each habitat within a specific buffer size.

Species richness is expected to be higher in forests or landscapes with a higher proportion of forest cover due to higher habitat complexity. In contrast, species abundance is expected to be higher in *bolanhas* or landscapes with a higher proportion of *bolanha* cover, given that field productivity may favour high reproductive rates in some species (Byrom et al., 2014; Foord et al., 2018). In addition, we expect the higher proportion of cashew orchards to have a negative effect both on species richness and abundance, given the well-studied impacts of these monoculture systems on several groups (Rege & Lee, 2023; Scales & Mardsen, 2008; Vasconcelos et al., 2015). Consequently, we expect species composition to be different among the three different habitats. In general, we expect a more evident habitat effect at the landscape scale related to the amount of each habitat type, given that the effect of habitat type at the local scale might be diluted due to the proximity between the various habitat types.

## 2. Methods

### 2.1. Study area

The study was conducted in the Oio region on northern Guinea-Bissau, between Farim and Mansaba cities (12º37’N, 15º20’W) (Figure 2.1). Sampling took place in the surroundings of five villages: Djalicunda (Dja), Bironqui (Bir), Bereco (Bere) and Demba So (Demb) in the Mansaba sector and Lenquebato (Lenq) in the Farim sector. The region has a smooth topography, with a maximum altitude of 77 m (WeatherSpark, 2023), and the landscape consists mainly of a mosaic of cashew orchards, rice paddies and forested areas. The country lies in the Northern Rainforest-Savanna Mosaic within the Rainforest Biotic Zone (Happold & Lock, 2013), characterised by a tropical climate with two seasons during the year, having an average annual temperature of 28.05ºC, and a mean annual precipitation of 1612.26 mm (Climate Change Knowledge Portal, 2023). The dry season occurs from November to May, in which the temperature ranges between 17.82 and 37.7ºC and the wet season from June to October, in which the temperature varies between 23.14 and 34.1ºC, with the highest precipitation between July and September (Climate Change Knowledge Portal, 2023). The precipitation patterns are subject to the seasonal movement of the Inter-Tropical Convergence Zone (ITCZ) (Lélé & Lamb, 2010), which makes the annual precipitation variability notable, with the wetter years linked to the northward shift of the ITCZ, determined by anomalies in the sea surface temperature (Nicholson et al., 2008).

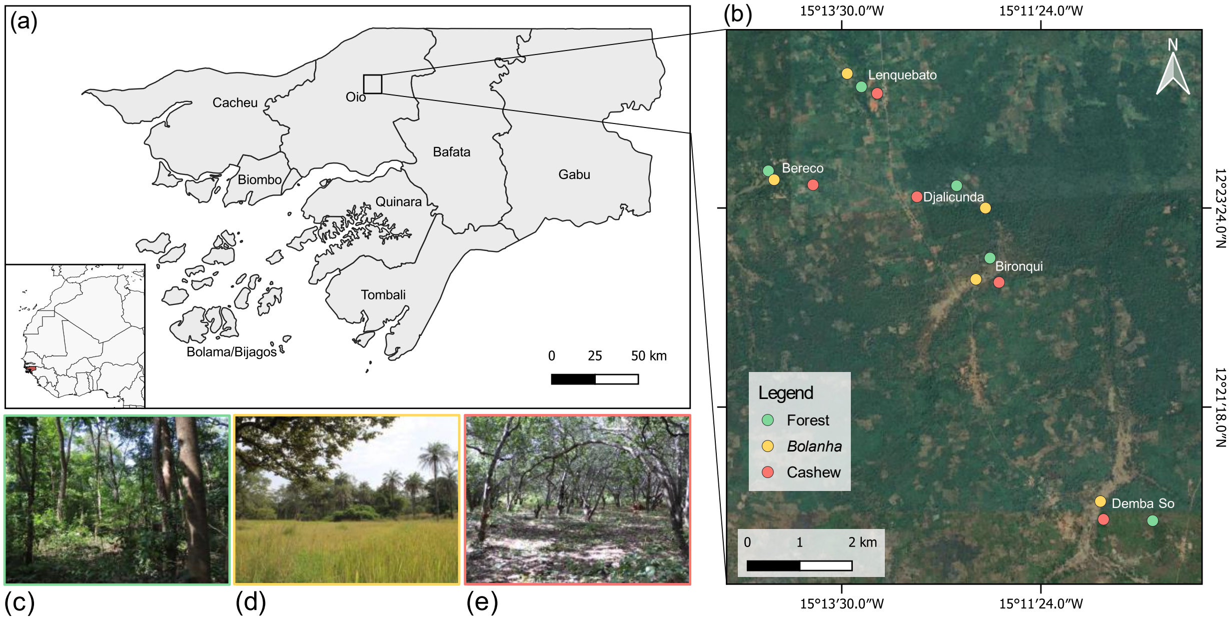

**Figure 2.1.** Study area and sampling sites; **(a)** Locations of Guinea-Bissau and the study area; **(b)** Study area map highlighting the locations of the 15 sampling sites, including three types of habitats — forest, *bolanha* and cashew (solid dots are colour-coded by habitat type) — across five villages: Lenquebato, Bereco, Djalicunda, Bironqui, and Demba So, in northern Guinea-Bissau. Each of the sampled habitat types is further illustrated by a picture: **(c)** forest, **(d)** *bolanha*, and **(e)** cashew orchards. Map sources: qGIS (2023) and Humanitarian Data Exchange (2021). Photos: Raquel Oliveira and Ana Filipa Palmeirim

In each village, three types of habitats were sampled, including forest (F), *bolanha* (B) and cashew orchards (C), amounting to a total of 15 sampling sites. Sampling was carried out on three occasions in 2022, one in June, another in October, and another in November-December. This allowed to maximise sampling sufficiency, while accounting for small mammal assemblage variation throughout the year. Forest habitats in Guinea-Bissau consist broadly of a mosaic of open forests and wooded savanna (Catarino et al., 2008). These habitats are typically dominated by trees without overlapping crowns at the highest stratum and with a consistent presence of shrub and herbaceous layers that are usually well-developed. Lianas are also frequently observed, extending to the tops of the tallest trees and creating a good connection network between trees. Tree species commonly found in dense forests or wooden savannas are also prevalent in open forests (Catarino & Indjai 2019, Jones, 1992). The sampled forests varied in patch size from approximately 0.129 km² and 3.51 km² and exhibited greater variation between each other compared to the remaining habitats: the average height of the five tallest trees in each forest ranged from 9.2 to 23 metres; tall grass cover ranged from 5 to 40%; one of the sampled forests (Bir-F) had significantly lower percentage of leaf litter (5%) than the overall values observed at other sites (mean ± SD = 67.5 ± 26.3%), also had a high percentage of bare ground (35%) compared to the overall values (mean ± SD = 1.25 ± 2.5%) and it was the only forest subject to slight flooding; both Bir-F and Lenq-F lacked a proper understorey with thin stems and lianas, which contributed to increase overall vertical complexity and connectivity between the forest floor and canopy. Each of the forest characteristics remained consistent over the three sampling occasions. Moreover, all the sampled forests are located near local communities, which use the forests particularly to extract forest products (such as fruit, bark, honey, and game). *Bolanhas* are flooded lowland rice ecosystems with scattered trees. The size of the *bolanha* patches varied from approximately 0.024 to 1.125 km². Cultivation begins with the removal of spontaneous herbaceous vegetation that grows yearly between rice crop cycles. Before the rice is sown, the soil is mobilised by hoes or a motor cultivator. Sowing is then carried out by burying the rice seed to avoid predation (A. Rainho, personal communication, 2023). In June, the surveyed *bolanhas* were dry and still needed to be cleansed before rice sowing, and some of them were being used as pastures by the cattle. In October, the *bolanhas* were flooded, and the rice was already growing, showing slightly different stages of growth across the different parcels. By December, the *bolanhas* were once again dry, and the rice was ripe and already being harvested. Cashew orchards consist of monocultures of the Brazilian-native species *A. occidentale,* typically composed of small land holdings that are locally owned. Trees are 2-3 metres tall and are spaced 4-5 metres apart. These cashew orchards are cultivated without the use of agrochemicals or irrigation (Catarino et al., 2015) and the patches sampled varied from 0.022 km² and 1.394 km² in size. In June, the trees were fruiting, and local people were often in the orchards to collect the cashew kernel. From October to December, the trees had no fruit. Additionally, in December, the undergrowth of some of the orchards was being cleared. Comprehensive details on the habitat characteristics can be found in dos Reis Silva (2023).

The present study involved the collection of samples from various sites under the ownership of several people from the villages sampled. Before sampling, we obtained permission from the landowners and each village's committee. This study is part of a broader project aiming to assess the potential contribution of bat and bird-mediated suppression of rice insect pests in Guinea-Bissau (EcoPestSupression, 2021). This research is being developed in collaboration with the local non-government organization *Federação Camponesa KAFO*, and the national authority in charge of managing biodiversity and protected areas, the *Instituto da Biodiversidade e das Áreas Protegidas* (IBAP).

### 2.2. Small mammal sampling

Small mammals were sampled during 3600 trap nights using one linear transect (100 m-length) per sampling site, consisting of ten Sherman traps (23 × 9 × 8 cm, H. B. Sherman Traps, Inc., Tallahassee, Florida), spaced by 10-m intervals. In the cashew orchards and *bolanha* transects, we placed all traps on the ground. In the forest transects, we placed the traps alternatively on the ground and in the understorey, except for Lenquebato and Bironqui, in which we placed them all on the ground since those forests mainly lacked a proper understorey connecting to the canopy. The traps placed in the understorey were set in places with links between tree branches or lianas to favour the capture of arboreal and scansorial species. Each trap was baited with a mixture of bananas, peanut butter, sardines and oatmeal and placed in the shade to prevent trapped animals from overheating.

In each of the three sampling seasons, traps remained active for eight consecutive nights and were checked daily during the morning since there were no significant records of diurnal species - only two species (*Arvicanthis* sp. and *Heliosciurus gambianus*), constituting 2.6% of the total sampled individuals, were known to be or to include diurnal activity (Happold, 2013; Happold & Happold, 2013; Monadjem et al., 2015). Captured individuals were measured and weighted. A tissue sample was also collected by making a standardised cut in the ear tip to be used in molecular analysis for species identification to the lowest taxonomic level possible, while allowing us to recognise subsequent recaptures. The molecular approach was employed due to the inherent challenges in field identification of small mammals. This study followed the Guidelines of the American Society of Mammalogists (Sikes 2016). Biological material collection and transport were carried out under the authorization from IBAP.

### 2.3. Molecular analysis

The collected tissue samples from the ear tips of each individual recorded were preserved in Eppendorf tubes containing 90% ethanol. These samples were stored and later analysed by specialized personnel at Centro de Investigação em Biodiversidade e Recursos Genéticos (CIBIO) of the University of Porto, Portugal. Genomic DNA was extracted from tissue samples following the ®BioBasic DNA Extraction Kit protocol with an overnight incubation. A gel electrophoresis analysis with 0.8% agarose stained with GelRed™ (Biotium) was used to assess the DNA quality and quantity.

The mitochondrial cytochrome b (Cytb) gene was used for molecular taxonomic identification, as widely used in other studies (e.g., Mamba et al., 2021; Lecompte et al., 2002; Pagès et al., 2010), and ensuring sufficient haplotype coverage for the studied families and genera. The Cytb gene was amplified using the primer set H15915/L14723 (Irwin et al., 1991), and a new set designed for this study, Cytb_RodF1 (forward, 5'- GAC ATG AAA AAT CAY CGT TG -3'), and Cytb_RodR1 (reverse, 5'- CAT TTY TGG TTT ACA AGA CC -3'), for the genera in which the former primer pair failed to amplify or to provide good quality sequences. PCRs (Polymerase Chain Reactions) were conducted in a final volume of 10 μL, corresponding to 5 μL of MyTaq™ Mix (Bioline), 0.4 μM of each primer, 3.2 μL of ultra-pure water, and 1 μL of DNA. The PCRs' thermal conditions used for the set H15915/L14723 were the following: initial denaturation at 95 ºC for 15 min; 7 cycles at 95 ºC for 30 s, 59 ºC decreasing 0.5 ºC per cycle for 30 s; 72 ºC for 1 min; and 33 cycles at 95 ºC for 30 s, 56 ºC for 30 s; 72 ºC for 1 min; and a final elongation step at 60 ºC for 10 min. The PCRs' thermal conditions used for the new primer set (Cytb_RodF1 and Cytb_RodR1) were the following: initial denaturation at 95 ºC for 15 min; 40 cycles at 95 ºC for 30 s, 53 ºC for 30 s; 72 ºC for 1 min; and a final elongation step at 60 ºC for 10 min. PCR products were purified with ExoSap (USB® ExoSAP-IT® PCR Product Cleanup, Affymetrix). A 10 μL sequencing reaction was prepared using BigDye® Terminator v3.1 Sequencing Kits (AB Applied Biosystems) following manufactures protocol. Finally, the sequencing products were purified with Sephadex and sequenced by capillary electrophoresis on an ABI 3500XL Genetic Analyzer (Applied Biosystems). The obtained chromatograms were validated using Geneious 10 v10.2.6 (Kearse et al., 2012) and sequences were identified to the lowest taxonomic level possible using the Basic Local Alignment Search Tool (BLAST) of the NCBI database (Sayers et al., 2022).

## 2.4. Landscape scale variables

In the absence of readily available land-use maps for the study region and due to challenges in automatic classification using remote sensing, particularly in distinguishing cashew orchards from forests due to significant spectral overlap (Pereira et al., 2022), we adopted a manual approach to classifying the landscape according to the different habitats under study (i.e., forest, *bolanha* and cashew orchards). With the visual aid of Google Earth imagery (Google Earth v7.3.6.9345, 2022) and ground-truth information collected during sampling, we carefully identified and marked areas corresponding to each habitat type. Other habitat types apart from the ones under study were not classified nor mapped. To quantify the proportion of each habitat to later be used in the landscape scale analyses, we established buffer polygons along each sampling transect with the following radius: 100, 150, 200, 250 and 300 m. These buffers served as spatial units for quantifying the proportions of each habitat type and were chosen taking into account the reported home ranges for some of the recorded species and genus while avoiding high buffer overlap of the adjacent sampling sites (*Mastomys erythroleucus*: 743-1625 m$^2$, Gebresilassie et al., 2006; *Praomys [tullbergi]*: 1,100-3,900 m$^2$, Happold, 1977; *Gerbilliscus guineae*: 1400– 1500 m$^2$, Happold, 2013; *Graphiurus [murinus]*: 2,514 m$^2$, Madikiza, 2010). Finally, for each of the buffer sizes, we calculated the proportion of each habitat type within each buffer by overlaying the habitat polygons with the buffer polygons along the sampling transects using the interception tool in QGIS [version 3.28.4] (QGIS Development Team, 2022). The resulting proportions are summarised in the Table 6.1. The percentage of classified habitat (comprising all three habitat types) within each buffer ranged from 35.49 to 97.99%.

## 2.5. Data analysis

All statistical analyses were conducted using R version 4.3.1 (R Core Team, 2023). The significance level was set to 0.05. One of the forest sampling sites (Dja-F) was identified as an outlier and subsequently excluded from all analyses. At this site, both observed species richness (1 species) and small mammal abundance (4 individuals) were significantly lower than the mean (± SD) values observed at any of the other sites (3.2 ± 1.5 species and 15.3 ± 10.9 individuals). We believe the particularly low values observed in Dja-F were due to the eventually lower effective sampling effort, given the number of traps found to be daily closed, and thus unlikely to have remained open the during the whole night. Indeed, during the second and third trapping sessions, more than 50% of all traps in Dja-F were found closed by the time of the daily checking, rendering the data collected at this site unreliable in reflecting the true ecological conditions and incomparable to the other sites where traps remained active throughout the sampling periods.

### 2.5.1. Local scale analysis

For a better visual interpretation of the species occurrence in each sampling site across all habitats, we created a matrix representing the presence-absence data using the “decostand” function of the R-package “vegan” (Oksanen et al. 2022). To analyse species composition across the different habitat types at a local scale, we employed a Non-Metric Dimensional Scaling (NMDS) using the Bray-Curtis distance and two dimensions (k = 2) using the “metaMDS” function from the “vegan” R package (Bray & Curtis 1957, Faith et al. 1987; Oksanen et al., 2022). The resulting NMDS ordination exhibited a stress value of 0.106. To assess the differences in species composition between the habitat types, we employed PERMANOVA (Permutational multivariate analysis of variance) via the “adonis” function of the "vegan" package (Oksanen et al., 2022). Pairwise differences were subsequently calculated using the “pairwise.adonis2” function from the “pairwise.adonis” package (Martinez Arbizu, 2020), with Bonferroni correction applied.

To assess how biodiversity metrics — species richness and Shannon diversity — vary across the different habitats at a local scale, we computed sample size-based extrapolation curves using the “iNEXT” package (Hsieh et al., 2016), within the framework of Hill numbers, as elucidated by Chao et al. (2014). Hill (1973) introduced Hill numbers as a group of diversity indices including different orders (*q*-values) that incorporates species richness and gives varying weight to abundance. This diversity metric consists of a single equation, which is defined as

$$^{q}D = \left(\sum_{i=1}^{S} p_i^{q}\right)^{1/(1-q)} \qquad \textbf{Equation 2.1}$$

where $S$ represents the number of species in the assemblage, and $p_i$ is the relative abundance of the $i^{th}$ species ($i = 1, 2, \ldots, S$). The parameter $q$ modulates the measure’s sensitivity to species relative frequencies. When $q = 0$, Equation 2.1 corresponds solely to the species richness, accounting for the effective number of species present irrespective of their relative abundances. For $q = 1$, Equation 2.1 is undefined, but its limits as $q$ tends to 1 yields the exponential of the well-known Shannon index, termed here as Shannon diversity:

$$^{1}D = \lim_{q \to 1} {}^{q}D = exp\left(-\sum_{i=1}^{S} p_i \; log\, p_i\right) \qquad \textbf{Equation 2.2}$$

The variable $^{1}D$ assigns weight to species proportionate to their occurrence frequency, giving more weight to species more frequently observed within the assemblage. The differences in species richness across habitats were assessed by visually comparing the rarefaction curves and their corresponding 95% confidence intervals. If the 95% confidence intervals of two habitats had no overlapping, we inferred that species diversity in those habitats was significantly different from each other. This framework using Hill numbers is now well established, having several advantages over the traditional diversity indices, including intuitive scaling, rarity-based expression, and explicit sensitivity to rare species (Roswell et al. 2021). Additionally, equalising coverage before measuring diversity can reduce bias in biodiversity assessments, done with the aid of the "extrapolation" function of the "iNEXT" package (Hsieh et al. 2016). This is particularly important in our study, given that the sampling sufficiency criterion was not met at certain sampling sites. This was assessed by generating rarefaction curves using the "rarecurve" function. This insight enabled us to estimate the expected diversity levels that could have been achieved with an increased number of captured individuals, thereby strengthening the robustness of our assessment (Figure 6.1).

### 2.5.2. Landscape scale analysis

At the landscape scale, we considered species abundance, in addition to observed and estimated species richness. Species abundance was defined as the sum of all individuals recorded in each sampling site, observed species richness was defined as the number of species present in a sampling site, and estimated species richness was calculated using a richness estimator (function: "ChaoRichness", package iNEXT (Hsieh & Chao, 2022)), based on the methods proposed by Chao (Chao, 1984; Chao, 1987). The strength at which the different biodiversity metrics are affected by each of the landscape variables might vary according to the spatial scale at which such relationship is looked upon. Thus, to examine the patterns of small mammal diversity (abundance, observed and estimated species richness) at a landscape scale, we first assessed whether there was a particular spatial scale at which these patterns were most notorious, the so-called 'scale of effect' (Jackson & Fahrig, 2015). To estimate the scale of effect, we considered the previously selected range of buffer sizes: 100, 150, 200, 250 and 300 m along each sampling transect.

We applied Generalised Linear Mixed Models (GLMMs) and Linear Mixed Models (LMMs) to determine the most appropriate scale of effect for each habitat type and response variable (abundance, observed and estimated species richness). For abundance, we fitted 15 GLMMs (three per habitat type × five buffer sizes) with a negative binomial distribution to account for data overdispersion. For observed species richness, we fitted 15 LMMs. Estimated species richness data was transformed using a logarithmic (base 10) transformation to meet the assumptions of normality required, allowing us to also fit 15 LMMs. In all models we included "village" as a random factor to account for potential spatial autocorrelation within villages, using the functions "glmer" and "lmer" functions in package "lme4" (Bates et al., 2015). Model performance was evaluated according to: (i) the Akaike Information Criteria corrected for small sample sizes (AICc) (Burnham & Anderson 2004) by, after ranking all models from lowest to highest AICc, considering a model to be the most plausible if the difference in AICc values between that model with lowest AIC and the model with the second lowest AIC was greater than 2; and, (ii) the conditional coefficient of determination ($R^2$, which represents the variance explained by the model, including both fixed and random effects) (Nakagawa et al. 2017). In cases in which the difference in the AICc of the two better ranked models was lower than 2, indicating that multiple models were equally plausible, we relied on the $R^2$ values to make our selection. The model with the highest $R^2$ value was consistently chosen as the preferred model among those with AICc values within this narrow range. This approach ensured that, when faced with a choice between similarly well performing models, we favoured the one that provided the highest degree of variance explained. The most parsimonious models were selected based on these criteria, ultimately revealing the spatial scale at which small mammal diversity variables exhibited the strongest relationship with the proportion of each of the habitat types within the study area (Table 6.2).

To examine the effect of each habitat type on small mammal diversity, we applied a GLMM on species abundance, fitted with a Negative Binomial distribution, and two separate LMMs, one for observed species richness and another for estimated species richness. Each explanatory variable — the proportions of forest, *bolanha* and cashew orchard within the landscape (buffer) — has been considered at their ‘best’ scale of effect. Prior to model fitting, we assessed multicollinearity among explanatory variables using variance inflation factors (VIFs). No significant multicollinearity was detected (all VIFs < 5), ensuring the independence of the explanatory variables in the models. Again, “Village” was included as a random factor in all models since it significantly improved the models' overall fit to the data (marginal $R^2$ = 0.61 vs. conditional $R^2$ = 0.91 for abundance; 0.39 vs. 0.96 for observed species richness and 0.37 vs. 0.96 for estimated species richness).

## 3. Results

A total of 230 small mammal individuals were captured across the 15 sampling sites, resulting in a capture success of 6.39% per trap-night (excluding 75 recaptures, which were not included in subsequent analysis). A total of 11 small mammal species were recorded, comprising nine rodents (Rodentia) and two shrews (Eulipotyphla), all of which being endemic to Africa (Figure 3.1a). The most commonly recorded species was the Guinea multimammate mouse (*Mastomys erythroleucus*) accounting for 53.9% of all captures, followed by Dalton's mouse (*Praomys daltoni*) with 30.0%. The most ubiquitous species was *P. daltoni*, being recorded in 14 out of the 15 sampling sites (Figure 3.1a). Four species were exclusively captured in the *bolanha* habitat: the grass rat (*Arvicanthis sp.*), the Rudd's mouse (*Uranomys ruddi*), the African giant shrew (*Crocidura olivieri*) and Hubert's multimammate mouse (*Mastomys huberti*), with the latter captured only once. The Cinderella shrew (*Crocidura cinderella*) was exclusively captured in the cashew habitat. The only species exclusively recorded (only once) in the forest was the Gambian sun squirrel (*Heliosciurus gambianus*) (Figure 3.1a). Within each habitat type, observed species richness ranged as follows: forests (1-4 species, mean ± SD: 2.50 ± 1.29, excluding Dja-F), *bolanhas* (3-6 species, 4.40 ± 1.41), and cashew orchards (2-5 species, 3.00 ± 1.41).

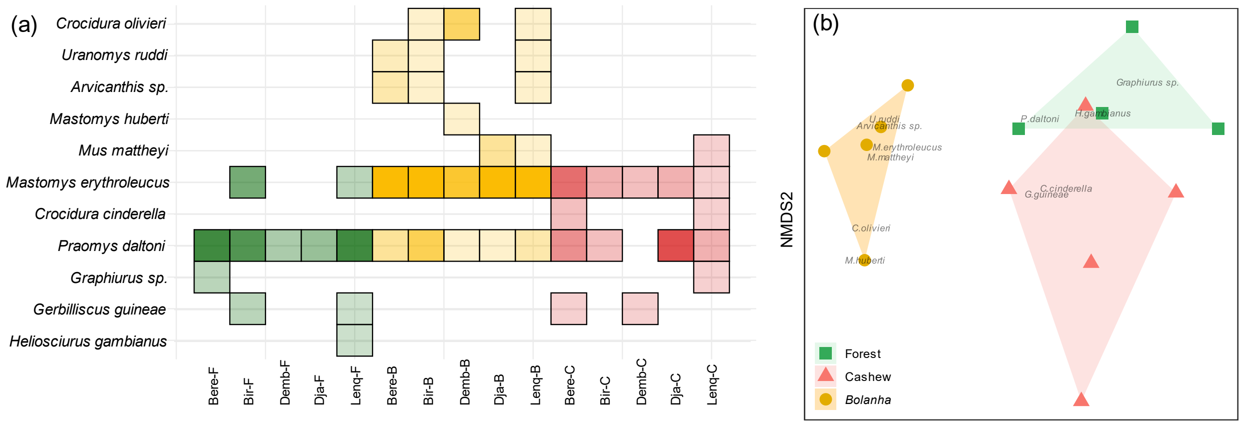

**Figure 3.1. (a)** Species recorded at each sampling site. Species incidence is denoted by a square that is colour-coded according to the habitat type (see legend). Each square's colour intensity corresponds to the corresponding relative frequency; **(b)** Non-Metric Dimensional Scaling (NMDS) depicting species composition (considering species abundance) across the three habitat types for 14 sampling sites (excluding the outlier Dja-F).

### 3.1. Local scale

Small mammal composition differed across the different habitats (PERMANOVA: $R^2$ = 0.52, P = 0.002, df = 2). Pairwise comparisons indicate that the *bolanha* assemblage exhibited significant differences from both forest ($R^2$ = 0.60, P = 0.009) and cashew ($R^2$ = 0.48, P = 0.009). Still, no differences were found between forest and cashew habitats ($R^2 = 0.21$, $P = 0.206$). The first axis of NMDS separates forest and cashew orchards from rice paddies, while the second axis of NMDS only slightly separates forest and cashew orchards (Figure 3.1b). The cashew site that is overlapping with the forest polygon is Dja-C: only two species were recorded in this habitat, the scansorial rodent *P. daltoni* and the terrestrial rodent *M. erythroleucus* (Figure 3.1a). The abundances of these species found in this site like those found in forest sites, likely contributing to higher similarity between them.

At a local scale, the individual-based accumulation curves revealed that species richness ($q = 0$) and Shannon diversity ($q = 1$) were similar across the three habitats, as evidenced by the overlapping 95% confidence intervals of extrapolated small mammal richness (Figure 3.2).

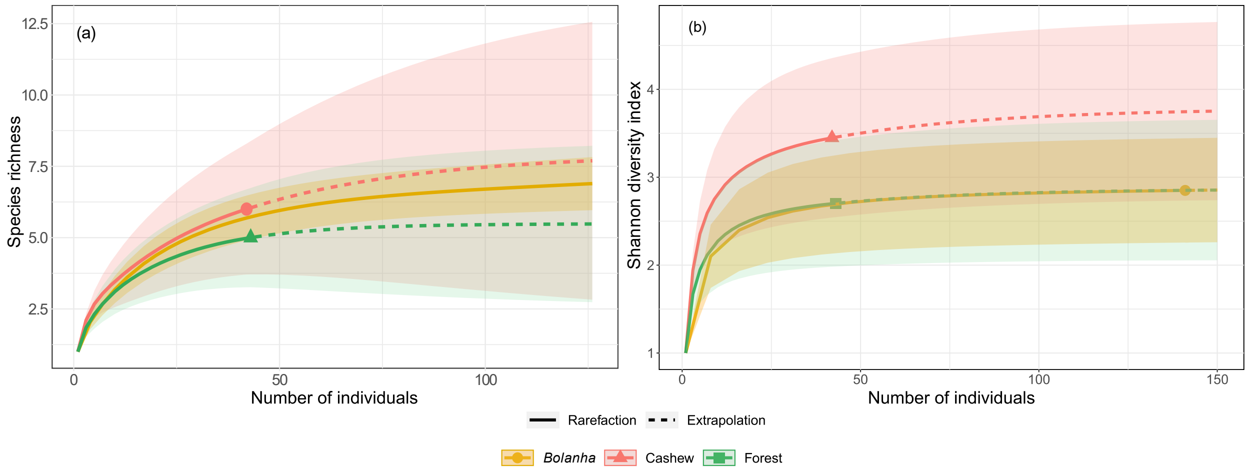

**Figure 3.2.** Individual-based extrapolation curves showing **(a)** species richness ($q = 0$) and **(b)** Shannon diversity ($q = 1$) surveyed across 14 sampling sites (excluding the outlier Dja-F) in northern Guinea-Bissau. Solid lines represent rarefaction, and dashed lines represent extrapolation. The solid dots, triangles and squares represent the reference samples, that is, the cumulative number of captured individuals at each habitat type. Shaded areas indicate 95% confidence intervals. The curves were extrapolated in (a) to the maximum sample size of 126 (triple the minimum observed sample size of 42, as suggested by Hsieh et al., 2016) and in (b) to 150 (just above the largest reference sample size of 141).

## 3.2. Landscape scale

Results for observed and estimated species richness were similar; therefore, the latter is only included in the supplementary material (Figure 6.2). The spatial scale at which the response variables exhibited the strongest relationship with the proportion of habitat differed among habitats and response variables. Regarding species abundance, the best scale of effect was at 100 m-radii for forest and cashew habitats and at 150 m-radii for the *bolanha* habitat. For observed species richness, the best scale of effect was at 100 m-radii for forest habitats, 150 m-radii for *bolanha* habitats, and 300 m-radii for cashew orchard habitats. Observed species richness was higher in landscapes with higher proportion of *bolanha* (Estimate = 1.03, $P \leq 0.001$). Small mammal abundance was also higher in landscapes with higher proportion of *bolanha* (Estimate = 0.41, $P = 0.008$), and lower under higher proportions of cashew orchards (Estimate = 0.34, $P = 0.032$) (Figure 3.3, Table 6.3).

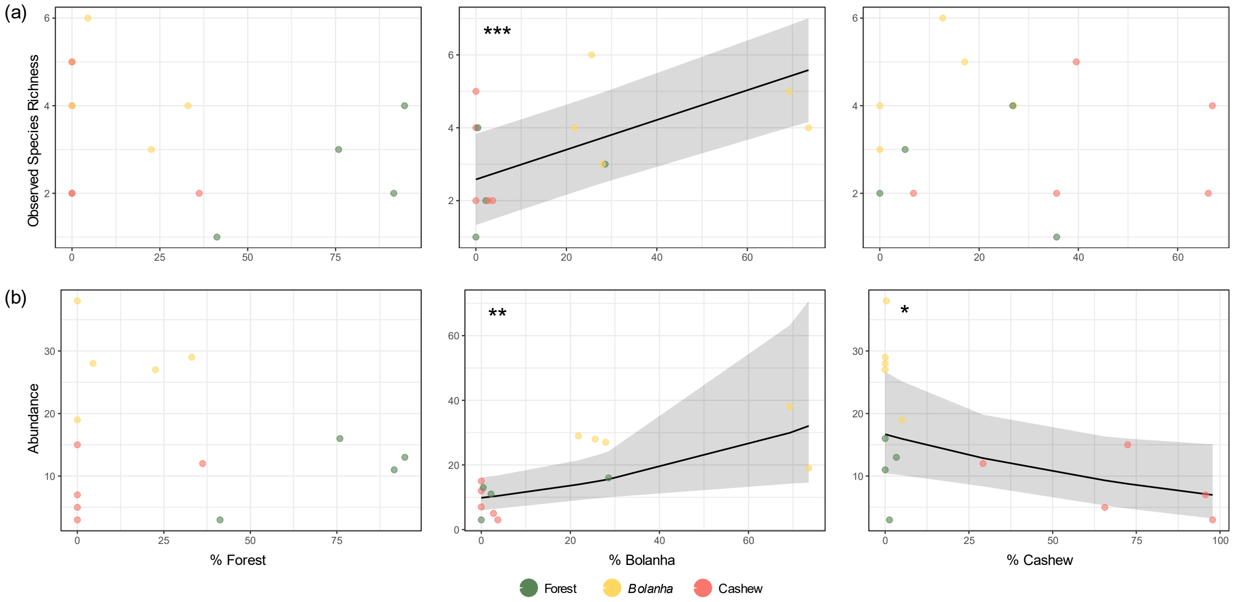

**Figure 3.3.** Relationships between the proportion of each habitat and **(a)** observed species richness and **(b)** species abundance of small mammals in northern Guinea-Bissau. Relationships were analysed using LMMs for observed species richness and GLMMs fitted with a negative binomial distribution for abundance. The relationships are only presented when they are significant and were fitted using the "ggmeans" function from the "emmeans" package. Significant relationships are indicated by the corresponding *P*-values symbols: * = $P \leq 0.05$; ** = $P \leq 0.01$; *** = $P \leq 0.001$. Shaded areas represent the 95% confidence regions. Data includes the 14 sampling sites (excluding the outlier Dja-F) nested within the five villages. Scale of effect: (a) observed species richness – Forest: 100m; Cashew: 300m; *Bolanha*: 150m; (b) abundance – Forest: 100m; Cashew: 100m; *Bolanha*: 150m.

## 4. Discussion

The need to meet human population increasing requirements while ensuring biodiversity conservation has challenged current management practices and called for nature-based solutions (Arroyo-Rodríguez et al., 2020; Laurance et al., 2013; Vogel et al. 2023; Tilman et al., 2011). In the present work, we assessed small mammal diversity patterns across different habitats in multiple mosaic-landscapes. At a local scale, rarefaction/extrapolation analyses did not reveal differences in species richness nor in the Shannon diversity index across the three habitat types. Species composition, however, differed between the *bolanha* habitat and the remaining habitat types. At a landscape scale, observed species richness was positively influenced by the proportion of *bolanha*, but not affected by forest cover as initially predicted. Moreover, small mammal abundance was also positively affected by the proportion of *bolanha*, a trend supported by other studies such as Byrom et al. (2014) and Foord et al. (2018). Additionally, the fact that abundance was also negatively affected by the higher proportion of cashew orchards corroborates existing studies highlighting the negative impacts of monoculture practices (Flynn et al., 2009; Hurst et al., 2013; Mamba et al., 2019; Scales & Mardsen, 2008; Vasconcelos et al., 2015).

### 4.1. Local scale

Small mammal composition differed across the three habitats: forest, *bolanha* and cashew orchards. In the forest habitat, we observed a dominance of the scansorial rodent *P. daltoni*, a species mostly associated with forests, but also frequently encountered near human settlements and cultivated areas (Happold, 2013; Monadjem et al., 2015), which is in accordance with our results, since it was also found in *bolanhas* and cashew orchards. The presence of the squirrel *H. gambianus*, a forest-dwelling species, was exclusively recorded in the forest habitat, supporting its forest preference. However, caution should be taken when speculating about this species, as it was captured only once, and the traps used are not optimal for capturing squirrels. The terrestrial rodent *Gerbilliscus guinea* and the arboreal *Graphiurus* sp. were found both in forests and cashew orchards, expressing their preference for wooded habitats (Happold, 2013; Monadjem et al., 2015). In the *bolanhas*, the terrestrial rodent *M. erythroleucus* was a prominent presence, although it was also found in the other habitats, but in a considerably lower abundance. This aligns with the generalist and ubiquitous habits of this species known to occur in various habitat types (Happold, 2013; Monadjem et al., 2015). Species from the *Mastomys* genus are considered crop pests, impacting various agricultural crops, including rice (planted and stored) (Funmilayo & Akande, 1977; Prakash, 2017). This species should be one to target if considering ecologically based rodent management in the future (Makundi et al., 1999; Singleton et al., 2021).

Additionally, several species were captured exclusively at the *bolanha* habitat: *C. olivieri*, *U. ruddi*, *Arvicanthis* sp. and *M. huberti*, species that are more commonly found in moist areas and/or habitats with abundant grasses (Abdel-Rahman Ahmed et al., 2008; Happold, 2013; Happold & Happold, 2013; Monadjem et al., 2015; Niang et al., 2022). The significant differences observed in small mammal composition between *bolanhas* and both forests and cashew orchards can be attributed to the pronounced differences in habitat structure (Mamba et al., 2019). *Bolanhas* are open areas with high grass cover and low structural complexity. They are characterised by seasonal fluctuations of flooding and consequent dry periods, constituting a dynamic and somewhat unstable environment. These conditions are conducive to attracting species with varying habitat requirements, including those preferring grasslands and open clearings, as well as more adaptable species capable of rapid adaptation to changing conditions.

In contrast, forests maintain a relatively stable and structurally complex profile with a consistent microclimate, making them suitable for species with more specific habitat requirements. Furthermore, while cashew orchards may not represent an ideal habitat for small mammals due to its apparent homogeneity (Temudo & Abrantes, 2014), they do share certain key features with forests. These include the presence of trees, which may serve as crucial habitat elements for certain species (like *P. daltoni* and *Graphiurus* sp.) (Happold, 2013; Monadjem et al., 2015), and the provision of shaded areas that mitigate extreme microclimatic conditions. Moreover, cashew orchards in Guinea-Bissau experience low management intensity, relying solely on rainfed agriculture and without the use of agrochemicals (Catarino et al., 2015). These attributes may explain the observed compositional similarities between cashew orchards and forests.

Interestingly, nearly all species found in cashew orchards are also present in the other habitats. This seems to indicate that the species present in this habitat are generalist and could also reflect the potential lower availability of resources within the cashew ecosystem, either structurally or trophically. This observation raises intriguing questions about the relationship between cashew orchards and the remaining habitats, with the former potentially functioning as a recipient habitat in a source-sink dynamic, as suggested by Rege et al. (2020). However, we found a species exclusive to cashew orchards: the rare and poorly known shrew *Crocidura cinderella*. The global information available on this species, as obtained from fewer than ten specimens, is very scarce. However, it had already been reported in the islands of Guinea-Bissau (Rainho & Palmeirim, 2018), but never in the continent. It was found in dry Acacia forests, grassland habitats and a banana plantation (Happold & Happold, 2013; Mittermeier & Wilson, 2018). Intriguingly, this species is rare but apparently generalist. Still, our single capture of this species in a cashew orchard aligns with its apparent generalist behaviour. However, our sampling effort was not exhaustive enough to conclude that this species does not occur in the other habitats. Nevertheless, all species found in cashew orchards are generalists, supporting the notion that monoculture plantations tend to favour generalist species (Hurst et al., 2013; Mamba et al., 2019; Rege & Lee, 2022; Vasconcelos et al., 2015). Over and above the more or less generalist habits of each species, the apparent general use of the different habitats across species underscores the importance of preserving mosaic landscapes to support a range of ecological niches essential to maintaining biodiversity.

Considering solely the habitat category, small mammal diversity metrics (species richness and Shannon diversity index) remained similar between the three habitats. In fact, there was quite a variance within each category, resulting in the wide 95% confidence intervals that overlap in the rarefaction/extrapolation curves, which may be attributed to several factors. Firstly, the proximity of the sampling transects to other habitat types was not taken into account when considering the habitat category solely. It may have induced cross-habitat influences: small mammals may use adjacent habitats, and their home ranges may overlap with multiple habitat types (Diffendorfer et al., 1995), further adding to the complexity of local scale diversity patterns, which is in accordance with our landscape scale results. Likewise, the proportion of the specific habitat type within a broader landscape and the habitats surrounding the sampled site (landscape composition) may have influenced the species assemblage within each habitat type (Carvalho et al., 2023; Mérő et al., 2015; William et al., 2023). Additionally, each habitat category was not entirely uniform, with slight variations in structure complexity, vegetation structure, likely resource availability and other ecological parameters between the various sampling sites of the same habitat. This heterogeneity within habitats may have contributed to the observed variance and the wide confidence intervals in our rarefaction/extrapolation curves. In summary, this result suggests the influence of other factors not considered when examining solely the habitat category (Arroyo-Rodríguez et al., 2020; Willis & Whittaker, 2002). For instance, we could detect the effect of more than one habitat types at the landscape scale.

### 4.2. Landscape scale

When considering the landscape scale and employing the ‘best’ scale of effect, our study revealed that observed species richness was positively influenced by the proportion of *bolanha* cover and not affected by forest cover. This result does not support the hypothesis that species richness would be higher in forests. We give two main potential explanations for this result. The primary explanation for this finding is likely linked to water availability. Unlike other habitats, *bolanhas* experience a seasonal flooding after the rainy season, providing a consistent water source that may be scarce in other areas. This availability of water in *bolanhas* fosters insect abundance (Oyediran et al., 1999), benefiting insectivorous and omnivorous small mammals such as shrews. Simultaneously, the presence of rice grain in these areas serves as resource for granivorous and omnivorous species. As such, for a significant part of the year, *bolanhas* become resource-rich habitats. Consequently, landscapes with a higher proportion of *bolanhas* are also richer in small mammal species and individuals. Secondly, it is important to acknowledge the state of the forests in our study area, which are not pristine and are actively used by local communities, exhibiting signs of degradation (Catarino et al., 2008; Mendonça, 2021). This may have contributed to the observed patterns, as the forests in our study may not harbour as many species nor individuals as expected due to their altered state and human disturbance (see section 4.3).

Regarding small mammal abundance at a landscape scale, our study highlighted the positive influence of *bolanha* cover. This observation aligns with previous research, which has demonstrated that greater food availability and water accumulation through a significant part of the year in agricultural crops provide ideal conditions for small mammal reproduction (Brown et al., 2017, Byrom et al., 2014; Niang et al., 2022). In contrast, cover by cashew orchards negatively influenced small mammal abundance, which is consistent with the findings of studies highlighting the potential drawbacks of cashew orchards on the abundance of several groups (Rege & Lee, 2023; Scales & Mardsen, 2008; Vasconcelos et al., 2015). It is noteworthy, however, that during fructification, cashew orchards can constitute important feeding areas as a supplementary food source for many species (Anderson et al., 2007; Hockings & Sousa, 2012; Rege et al., 2020; Visalberghi et al., 2016). However, that has never been verified for small mammals. This negative influence of cashew orchard cover on small mammal abundance can thus be attributed to the lower resource availability in this habitat, such as food, water and shelter. Additionally, the lower abundance on this habitat supports the previously stated notion these habitats may act as potential "habitat extensions" of nearby forests and *bolanhas*.

Furthermore, it is essential to recognise that the study area's landscape is highly heterogeneous, characterised by a complex interweaving of different habitat types, such as forests, *bolanhas*, and cashew orchards, close to each other. As a result, "edge effects", which refer to the influences of transitional zones between different habitat types have on animal assemblages (Murcia, 1995), are relatively large and pervasive throughout the landscape. In contrast, the areas within each habitat where typical environmental conditions remain relatively undisturbed by these transitional zones, the 'core' of each habitat, are limited in extent and influence, resulting in diminished ecological significance (Bender et al., 1998). This makes it challenging to untangle the specific impacts of each habitat type in small mammal communities and can lead to more homogeneous communities and potentially reduce overall beta diversity (Ehlers Smith et al., 2020). Future studies could delve deeper into the intricacies of these edge effects and their potential impact on species diversity patterns in mosaic landscapes. The small size of habitat patches within the landscape, combined with the degraded state of our study area's forests, probably explains the lower number of species found compared to what we were expecting, a trend also verified in Rossinyol (2023). The richness and abundance patterns observed underscore the importance of considering both ecological history and landscape context when evaluating the impacts of land use on biodiversity (Carvalho et al., 2023; Ehlers Smith et al., 2020). These findings emphasise the complex interactions between small mammal communities and different habitat types within mosaic landscapes, emphasising the need for a more comprehensive understanding of the landscape as a whole, rather than confining analyses solely to local habitat categories (Arroyo-Rodríguez et al., 2020; Willis & Whittaker, 2002).

### 4.3. Limitations and further studies

It is essential to acknowledge the caveats of our study, notably the potential sampling insufficiency. The rarefaction curves (Figure 6.1) for *bolanha* reach an asymptote, indicating that our sampling effort in this habitat was likely sufficient to capture a significant portion of the species diversity present for it to be representative of the actual species richness in that habitat. Conversely, the rarefaction curves for forest and cashew habitats do not reach an asymptote, suggesting that our sampling effort in these habitats may not have captured the full extent of species present. Several factors could have contributed to this, including the potentially lower small mammal densities in these habitats, which makes it more difficult to capture enough individuals to reach the number of species closest to reality, or the probable higher structural complexity of these habitats compared to *bolanha*, making thorough sampling more challenging. Consequently, when comparing species richness between *bolanha* and either forest or cashew, we may have underestimated the species richness in these two habitats. This underestimation may influence the interpretation of differences in species richness between these habitats, particularly if the differences are not statistically significant, as is observed in our study. To address this limitation, we employed a richness estimator, which yielded results consistent with observed species richness (Figure 6.2). However, it is worth noting that this does not entirely eliminate the potential for underestimation.

One of the limitations of this study is the relatively low replication at the site level. We sampled a total of five sites within each habitat category, which, although representative of the study area, constitutes a limited number of replicates. Low replication can introduce variability and limit the generalizability of our findings. Ideally, a larger number of sampling sites would provide a more comprehensive representation of the small mammal community within each habitat type and enhance the statistical power of our analyses. However, due to logistical constraints and resource limitations, we were restricted in our ability to increase site replication. Researchers should thus bear in mind that the outcomes of this study may be influenced by the restricted number of sampling sites, and caution should be exercised when extrapolating these findings to broader contexts.

Another limitation of this study relates to the intrinsic constraints of the methods utilised, specifically not accounting for closed traps and recaptures. The higher number of recaptures observed in *bolanhas* may lead to an underestimation of the abundance of small mammals in that habitat. This is because traps occupied by previously captured individuals were not available for capturing additional ones. Consequently, such an underestimation would likely exacerbate the already significant differences in abundance between the *bolanha* habitats and others.

Additionally, it is important to address certain considerations regarding the spatial scale employed in our study. The spatial scale, which we determined as the scale of effect for abundance and observed species richness, varied among habitats and ranged between 100-300 m. Notably, for observed species richness in the cashew habitat, we identified the scale of effect to be 300 m, representing the largest scale analysed. As suggested by Jackson & Fahrig (2015), the accurate scale of effect may extend beyond our maximum scale analysed. Consequently, we may have missed an eventual significant effect of cashew orchards on species richness at a broader scale, not considered here. Our study's design and the distance between sampling sites limited our ability to explore these larger scales without compromising the spatial independence of our data points. Nonetheless, it is pertinent to underscore that determining the scale of effect serves simply as an intermediate step, facilitating our landscape scale analysis by appropriately accounting for habitat proportions. It does not constitute a primary outcome of our research. Therefore, our conclusions are valid within the spatial scales we examined. Still, we acknowledge the need for future research to investigate the ecological impacts of cashew orchards at broader spatial scales. This could provide valuable insights into the scale-dependent nature of these impacts and their implications for biodiversity conservation.

Certain environmental and ecological parameters such as the size of habitat patches, the presence of other land-use types and the edge density were not accounted for in our study. However, we acknowledge that these parameters may influence the observed patterns. To address this, we plan to incorporate them into our future research and outputs of this work.

Furthermore, the interpretations made in this study are based on observed patterns and warrant further investigation. Future research could delve deeper into the microclimatic variables and structural elements driving these differences in small mammal composition. Additionally, examining the ecological roles of individual species within each habitat type may provide a profound understanding of their contributions and disservices to ecosystem functioning and conservation in this multifaceted landscape.

### 4.4. Implications for conservation

Our study serves as a baseline knowledge for understanding small mammal diversity patterns in a mosaic landscape so that it is possible to develop tailored management strategies and policies aimed at enhancing crop productivity and food security while concurrently addressing pest-related challenges and preserving biodiversity. Integrating mosaic landscapes as a benefit-sharing mechanism that supports biodiversity conservation through the maintenance of livelihood sustenance can promote improved coexistence and may help stop the devastating trend towards adopting monoculture practices and other ecologically damaging practices (Arroyo-Rodríguez et al., 2020; Benton et al., 2003; Fahrig et al., 2011; Fulgence et al. 2022, Reynolds et al., 2018; Tscharntke et al., 2012). This increase in landscape heterogeneity is valuable from an ecological perspective and a safeguard for local communities to not rely solely on monocultures as a source of income, as they are more vulnerable to pests and diseases (Matson et al. 1997; Laurance et al. 2013; Tilman et al., 2011). Nevertheless, while mosaic landscapes appear to be beneficial for small mammals, that might not be the case for other taxa. Certain larger-bodied mammals, for example, may not meet the necessary requirements in a mosaic landscape made of small patches of different habitat types (Laurance & Vasconcelos, 2004). This was observed by Rossinyol (2023) in this very landscape, where species richness was expected to be significantly higher. Nonetheless, Rege et al. (2020) demonstrated that some terrestrial mammals use cashew orchards in a mixed forest-cashew landscape, suggesting that even larger mammals can benefit from mosaic landscapes. Mosaic landscapes comprising agricultural activities and agroforestry systems have proved that they can maintain a considerable proportion of the expected diversity and contribute to the conservation of several taxa, such as bats (Carrasco-Rueda & Loiselle, 2020; Harvey & González Villalobos, 2007), birds (Berg, 2002; Carrasco-Rueda & Loiselle, 2020; Harvey & González Villalobos, 2007; Reynolds et al., 2018) and herpetofauna (dos Reis Silva, 2023; Fulgence et al., 2022). This is verified since diverse vegetation types within mosaic landscapes can provide nesting sites, food sources, and suitable foraging habitats for different species, thus boosting the overall regional diversity. In any case, retaining large patches of native forests or restoring the degraded natural habitat features within production landscapes to improve overall landscape quality is crucial for maintaining diversity for all taxa (Carrasco-Rueda & Loiselle, 2020; Ehlers Smith et al., 2020; Harvey & González Villalobos, 2007; Mérő et al., 2015) and reducing the negative impacts of land-use changes, such as cashew expansion, as evidenced by other studies (Hurst et al., 2013; Vasconcelos et al., 2015). More specifically, our study shows that to maximise small mammal diversity within these mosaic-like landscapes, maintaining a reasonable proportion of *bolanha* and minimizing the proportion of cashew orchards appears to be a crucial strategy.

It is also essential to consider not only the general diversity patterns but also the identity of the species sampled. Focusing solely on richness and abundance may overlook crucial nuances in ecological communities, such as species turnover dynamics. For example, our results showed that the small mammal community in *bolanha* habitats was distinct from the communities in forest and cashew orchards, even though the overall species richness was similar in all three habitats. Additionally, the species present in the cashew orchards were generalist, suggesting that the cashew orchards may be less suitable for small mammals than the other two habitats. This suggests that the identity of the species present in each habitat is more important than simply the number of species present. Understanding which species are present, absent, or undergoing turnover, especially concerning specialists, generalists, and invasive species, can provide critical insights regarding ecosystem health. By acknowledging the identity of the species sampled, we gain a deeper understanding of the underlying ecological processes, facilitating more informed and targeted conservation strategies that aim to preserve not just quantities but the quality and resilience of biodiversity.

For example, while rodent pests are species to target to improve crop production, insectivorous species such as shrews play an important role in ecosystem functioning by helping to regulate arthropod pest populations naturally. The conservation of these insectivores can mitigate the reliance on harmful pesticides and their impacts on biodiversity and human health, ensuring a more sustainable and balanced agricultural system.

Moreover, we strongly believe that conservation can only be made with local people. Conservation efforts should involve the participation and collaboration of local communities and stakeholders with a direct interest and influence on environmental issues. Conservation with people also implies that the social, cultural and economic dimensions of conservation are taken into account, including food security and resource provision for the local communities. A way to guarantee food security and enhance agricultural production while safeguarding biodiversity can be achieved through ecological intensification strategies that encompass ecosystem services and disservices and nature-based solutions in agricultural management (Pender, 1998; Pimentel et al., 1992; Tscharntke et al., 2012). Our findings also contribute to the establishment of a robust base of knowledge for the species present within the study area, laying the groundwork for the application of ecologically based rodent management strategies, such as those proposed by Brown et al. (2017), Donga et al. (2022) and Singleton et al. (2021).

As a heartfelt gesture of appreciation and a way to give back to the communities that welcomed us into their midst during this study, we created a Photographic Guide to Small Mammals of Northern Guinea-Bissau (Supplementary Photographic Guide to Small Mammals of Northern Guinea-Bissau). This guide is not just an outcome of our study but also a tribute to the support and cooperation we received from the local people. We aim that this visual guide will serve as an educational resource, by offering a window into the fascinating world of small mammals that inhabit this region, hoping to strengthen the bond between wildlife and the local communities.

## 5. References

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# 6. Supplementary material

**Table 6.1.** Proportion of each habitat within each buffer (100, 150, 200, 250 and 300 m) across the 14 sampling sites (excluding Dja-F) in northern Guinea-Bissau. %F = percentage of forest habitat in the buffer considered; %C = percentage of cashew in the buffer considered; %B = percentage of *bolanha* in the buffer considered

| | | Forest | | | | Cashew | | | | | Bolanha | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Bere-F | Bir-F | Demb-F | Lenq-F | Bere-C | Bir-C | Demb-C | Dja-C | Lenq-C | Bere-B | Bir-B | Demb-B | Dja-B | Lenq-B |
| **100 m** | **%F** | 91.57 | 75.88 | 41.24 | 94.65 | 0.00 | 0.00 | 0.00 | 36.21 | 0.00 | 33.02 | 0.00 | 0.00 | 22.57 | 4.51 |
| | **%C** | 0.00 | 0.00 | 1.28 | 3.34 | 72.42 | 65.59 | 97.80 | 29.23 | 95.71 | 0.00 | 0.35 | 5.01 | 0.00 | 0.00 |
| | **%B** | 0.00 | 9.92 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 31.83 | 90.32 | 81.97 | 48.72 | 35.62 |
| **150 m** | **%F** | 78.40 | 60.88 | 33.01 | 78.01 | 0.00 | 0.00 | 0.00 | 41.13 | 0.00 | 37.02 | 0.00 | 0.00 | 50.54 | 9.91 |
| | **%C** | 0.00 | 0.00 | 6.49 | 10.17 | 67.37 | 44.82 | 87.94 | 21.57 | 71.70 | 0.00 | 8.05 | 14.85 | 0.00 | 0.00 |
| | **%B** | 2.20 | 28.58 | 0.00 | 0.44 | 0.00 | 2.75 | 3.71 | 0.00 | 0.00 | 21.80 | 69.25 | 73.52 | 27.93 | 25.58 |
| **200 m** | **%F** | 59.50 | 47.93 | 30.25 | 61.57 | 0.00 | 0.00 | 0.00 | 44.01 | 1.53 | 39.32 | 0.00 | 0.00 | 65.88 | 20.11 |
| | **%C** | 0.00 | 0.70 | 17.52 | 16.21 | 65.78 | 33.20 | 80.18 | 14.18 | 50.50 | 0.00 | 13.69 | 20.92 | 0.00 | 0.92 |
| | **%B** | 7.64 | 38.61 | 0.00 | 1.73 | 0.00 | 8.96 | 12.06 | 0.00 | 0.00 | 16.59 | 58.12 | 69.59 | 19.06 | 16.77 |
| **250 m** | **%F** | 46.29 | 40.13 | 29.41 | 50.10 | 0.00 | 0.00 | 0.00 | 48.88 | 8.85 | 35.90 | 0.00 | 0.00 | 71.76 | 25.76 |
| | **%C** | 0.00 | 2.61 | 27.88 | 20.59 | 66.29 | 33.09 | 72.28 | 9.48 | 41.45 | 0.00 | 15.63 | 24.56 | 0.00 | 5.93 |
| | **%B** | 9.29 | 43.02 | 0.00 | 5.40 | 0.00 | 13.61 | 18.31 | 0.00 | 0.00 | 13.55 | 51.61 | 67.17 | 15.63 | 11.67 |
| **300 m** | **%F** | 36.71 | 35.80 | 29.11 | 43.22 | 0.00 | 0.00 | 0.00 | 51.59 | 15.66 | 32.15 | 0.11 | 0.00 | 72.21 | 29.20 |
| | **%C** | 0.00 | 5.09 | 35.63 | 26.78 | 67.00 | 35.61 | 66.18 | 6.78 | 39.58 | 0.00 | 17.10 | 26.97 | 0.00 | 12.70 |
| | **%B** | 9.00 | 43.73 | 0.00 | 6.01 | 0.00 | 18.15 | 21.33 | 0.00 | 0.00 | 11.41 | 47.28 | 64.69 | 13.79 | 8.73 |

**Table 6.2.** Model performances according to the Akaike Information Criteria corrected for small sample sizes (AICc) and the conditional coefficient of determination (R-squared, which represents the variance explained by the complete model - fixed and random effects). For abundance, 15 GLMMs (three per habitat type × five buffer sizes) were fitted with a negative binomial distribution. For observed and estimated species richness, 15 LMMs were fitted for each.

| | | 100 | | 150 | | 200 | | 250 | | 300 | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | AICc | $R^2$c | AICc | $R^2$c | AICc | $R^2$c | AICc | $R^2$c | AICc | $R^2$c |
| Forest | Abundance | **114.7778** | **0.026322** | 115.0854 | 0.000893 | 114.9788 | 0.008934 | 114.8712 | 0.015825 | 114.8523 | 0.016601 |
| | Observed S | **65.34437** | **0.525216** | 65.33467 | 0.485652 | 65.71321 | 0.436886 | 65.87927 | 0.425445 | 66.02886 | 0.418639 |
| | Estimated S | 27.17857 | 0.588062 | **27.05013** | **0.560921** | 27.17751 | 0.542165 | 27.25393 | 0.55085 | 27.38918 | 0.560015 |
| Cashew | Abundance | **107.9924** | **0.469505** | 108.8562 | 0.426291 | 109.7039 | 0.37443 | 110.0422 | 0.347127 | 110.2771 | 0.329618 |
| | Observed S | 68.18975 | 0.241427 | 67.80354 | 0.229627 | 67.41828 | 0.229969 | 67.18884 | 0.236237 | **67.09594** | **0.245151** |
| | Estimated S | 30.42914 | 0.315408 | 29.92678 | 0.320883 | 29.53055 | 0.331728 | 29.32132 | 0.333814 | **29.20867** | **0.335502** |
| *Bolanha* | Abundance | 100.473 | 0.89118 | **98.79931** | **0.929878** | 105.0268 | 0.952554 | 109.7648 | 0.866004 | 112.5105 | 0.795721 |
| | Observed S | 57.75627 | 0.840548 | **57.40376** | **0.855818** | 60.13607 | 0.819902 | 62.32295 | 0.765859 | 63.39216 | 0.732963 |
| | Estimated S | 21.50415 | 0.817627 | **20.76161** | **0.840371** | 22.04905 | 0.835304 | 23.55426 | 0.810091 | 24.38982 | 0.796514 |

**Table 6.3.** GLMM and LMMs outputs for the final models. For each biodiversity metric, each landscape variable has been considered at its best scale of effect.

| | **Estimate** | **Std. error** | **df** | **t value** | **Pr (>\|z\|)** | |
|---|---|---|---|---|---|---|
| **Species abundance** | | | | | | |
| F100 | -0.1752 | 0.1246 | 6.187420 | -1.406 | 0.15973 | |
| C100 | -0.3430 | 0.1605 | 6.410596 | -2.137 | 0.03260 | * |
| B150 | 0.4060 | 0.1528 | 6.297316 | 2.657 | 0.00788 | ** |
| **Observed species richness** | | | | | | |
| F100 | -0.3111 | 0.1572 | 6.1874 | -1.980 | 0.093630 | . |
| C300 | 0.2401 | 0.1952 | 6.4106 | 1.230 | 0.261923 | |
| B150 | 1.0302 | 0.1625 | 6.2973 | 6.338 | 0.000597 | *** |
| **Estimated species richness** | | | | | | |
| F150 | -0.04896 | 0.03445 | 6.04782 | -1.421 | 0.204691 | |
| C300 | 0.08129 | 0.04128 | 6.19841 | 1.969 | 0.094943 | . |
| B150 | 0.21846 | 0.03221 | 6.14798 | 6.783 | 0.000453 | *** |

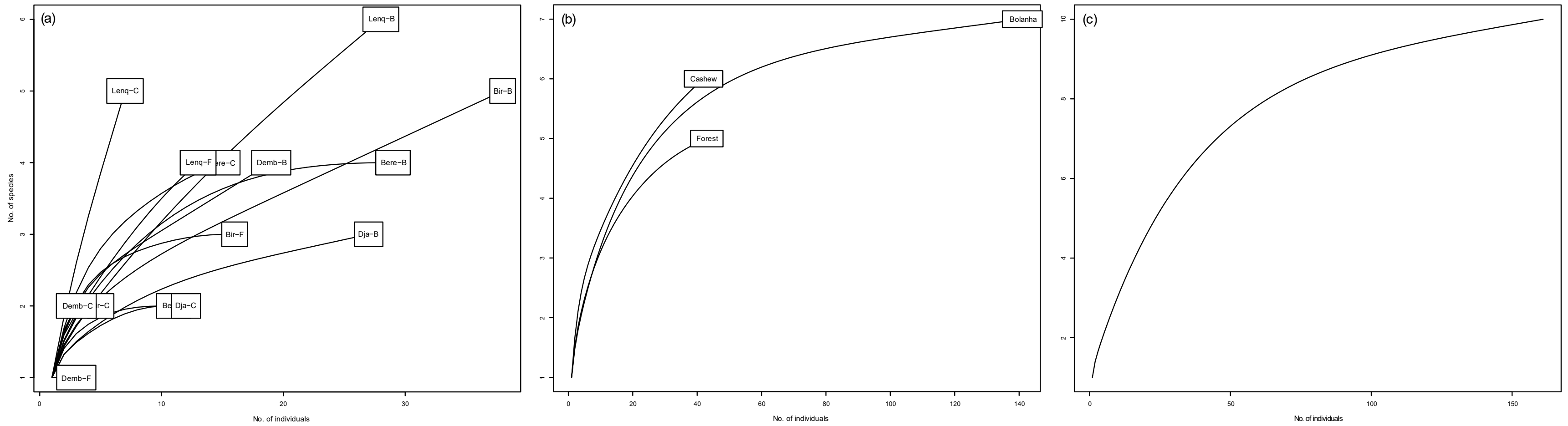


**Figure 6.1.** Individual-based accumulation curves for small mammals in north Guinea-Bissau across **(a)** sampling sites, **(b)** habitat type and **(c)** study area.

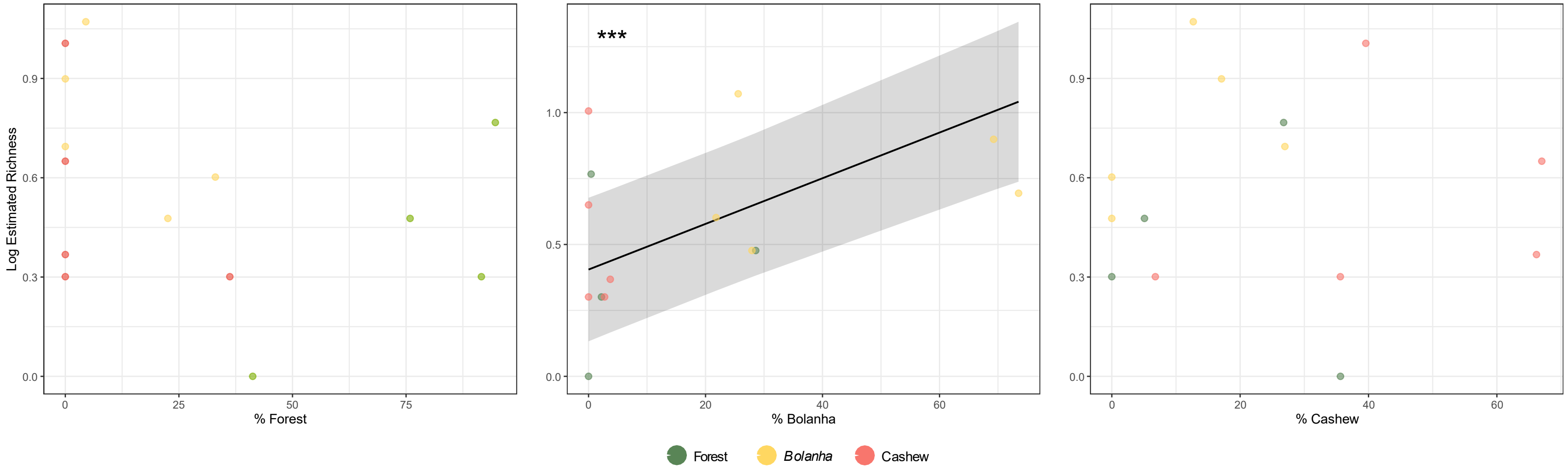


**Figure 6.2.** Relationships between the proportion of each habitat and estimated species richness of small mammals in northern Guinea-Bissau. They were estimated using LMMs. The relationships are only presented when they are significant and were fitted using the "ggmeans" function from the "emmeans" package. Significant relationships are indicated by the corresponding *P*-values symbols: * = $P \leq 0.05$; ** = $P \leq 0.01$; *** = $P \leq 0.001$. Shaded areas represent the 95% confidence regions. Data are presented for the 14 sampling sites (excluding the outlier Dja-F) nested within the five villages. Scale of effect for estimated species richness – Forest: 150m; Cashew: 300m; *Bolanha*: 150m

## 7. Supplementary Photographic Guide to Small Mammals of Northern Guinea-Bissau

# GUIA FOTOGRÁFICO DE
# PEQUENOS MAMÍFEROS
# DO NORTE DA GUINÉ-BISSAU

Raquel Oliveira, Ana Filipa Palmeirim, Ana Rainho, Djunco Dabo, Patrícia Guedes

# OBJETIVO E MÉTODOS

### *Português*

Este estudo teve como objetivo conhecer quais as espécies de pequenos mamíferos presentes na paisagem mista floresta–caju–bolanha na região de Oio, na Guiné–Bissau.

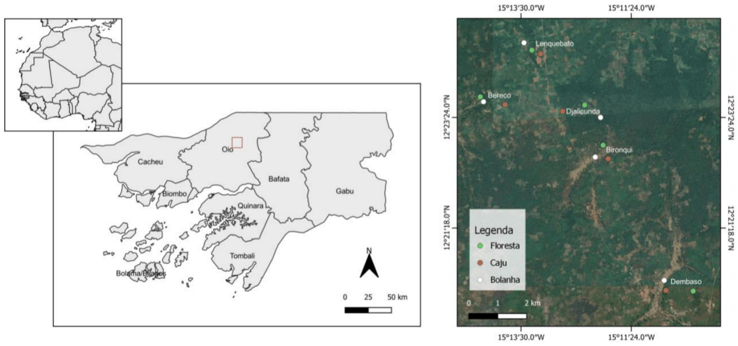
A recolha de dados decorreu nos meses de junho, outubro/novembro e dezembro de 2022 em 5 tabancas: Djalicunda, Bironqui, Bereco, Demba So e Lenquebato. De forma a capturar os animais, foram colocadas várias armadilhas nos diferentes locais, que foram iscadas com uma mistura de manteiga de amendoim, banana, sardinha e aveia. Os indivíduos capturados foram marcados e identificados, e após a medição dos mesmos foram novamente libertados.

### *Crioulo*

És studu tené suma objectivo kunsi kal ki diferentes tipus di limarias piquininus ku stá nas és diferentes kaus, mato, horta de cadju ku bulanhas na region de Oio na Guiné–Bissau.

Tarbadjus di buska dadus di és studu fasidu na mises di Junho, Outubro, Novembro ku Dezembro di ano 2022 na tabancas di: Djalicunda, Bironqui, Bereco, Demba–só e Lenquebato. Pa pudi panha kil limarias pikininus i pudu manga di armadilhas na diferentes kaus ndé ku i pudu isca ku purparadu ku mantega di mancara, banana, sardinha ku aveia. Limarias ku panhadu é marcadu i é identificado, é mididu dipus é largadu.

Tudu traduções na és guia i fasidu pa Mutaro Camará.



## HABITATS

### Floresta

Presente na floresta

Ausente na floresta

### Caju

Presente no caju

Ausente no caju

### Bolanha

Presente na bolanha

Ausente na bolanha

3

## O DECORRER DO TRABALHO

4

# RATO MULTIMAMADO DA GUINÉ

## *Mastomys erythroleucus*

Djalicunda Bironqui Bereco Demba So Lenquebato

És I tipo de animal mas tchiu na bulanhas ku studadu, mas é ta konsihui també vivi na manga di utrus kaus

Esta foi a espécie mais abundante nas bolanhas estudadas, mas consegue sobreviver numa grande variedade de habitats

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# RATO DE DALTON

## *Praomys daltoni*

Djalicunda Bironqui Bereco Demba So Lenquebato

Esta espécie vive tanto no chão como no estrato arbóreo, sendo encontrada frequentemente em árvores

És tipu di animal ta vivi na tchon ku zonas de arvuris, I mas ta incontradu na arvuris

# MUSARANHO GIGANTE AFRICANO

Rato djidiu (crioulo); Nhina solo (mandiga)

*Crocidura olivieri*

Djalicunda Bironqui Bereco Demba So Lenquebato

Alimenta-se de invertebrados, como formigas, escaravelhos, millípedes, térmitas e aranhas

É ta kumé bitchus suma, firmingas, Rola, Pipa, cem-pé, Bagabaga, ku Aranhas

# RATO DE MATTHEY

*Mus mattheyi*

Djalicunda Bironqui Bereco Demba So Lenquebato

Rato de terra pikininu, mangadel associadu ku zonas di savana, mas també ta odjadu na zonas k pekaduris esta nél

Ratinho terrestre muito pequeno. Maioritariamente associado a habitats de savana mas também é encontrado em ambientes humanos

# GERBIL GUINEENSE

## *Gerbilliscus guineae*

Djalicunda Bironqui Bereco Demba So Lenquebato

És tipu di animal I di terra I ta ianda I salta. I ta fassi cobas fundus, I ta sukundi entrada di coba k manga de reia e padja

Esta espécie é terrestre e desloca-se a saltitar. Escava tocas relativamente profundas e complexas cuja entrada é escondida por montes de terra

# RATO DE PRADO

## *Arvicanthis sp.*

Djalicunda Bironqui Bereco Demba So Lenquebato

A sua atividade é maioritariamente diurna, ao contrário dos restantes ratos que são mais ativos de noite

Sé atividades I mas ta cedu didia, ao contrário di kil utrus ratos ki I dinote

# RATO DE RUDD

Nhina uleu (mandiga)

*Uranomys ruddi*

Djalicunda Bironqui Bereco Demba So Lenquebato

Esta espécie tem patas fortes e grossas e usa-as para escavar tocas. Alimenta-se maioritariamente de insetos

És tipu di animal tene patas forti i grós ki ta ussa pa coba tchon, i manga delis ta cumé bitchu

# MUSARANHO CINDERELA

Rato djidiu (crioulo)

*Crocidura cinderella*

Djalicunda Bironqui Bereco Demba So Lenquebato

És tipu di animal I muito difícil, I na mundo I ka muito kunsidu. També I mas piquinino di ki kil utrus animais ku panhadu

Espécie muito rara e mundialmente pouco conhecida. É também a mais pequena das espécies capturadas

# ARGANAZ AFRICANO

Saninho de cima (crioulo)

*Graphiurus sp.*

Djalicunda Bironqui Bereco Demba So Lenquebato

Espécie arborícola que se assemelha a um pequeno esquilo. Precisam de alguma estrutura (árvores, rochas, casas) para fazerem os seus ninhos

Animal di padja ku ta parci, i ta parci ku saninhu piquinu. É ta pricisa de arvoris, pedras ku casas pa pudi fassi sé ninhus

# RATO MULTIMAMADO DE HUBERT

*Mastomys huberti*

Djalicunda Bironqui Bereco Demba So Lenquebato

Assim suma purmeru animal des guia, es limarias I kunsidu papia di manga di pares di mama pa pudi suporta manga di fidjus

Tal como a primeira espécie neste guia, estes animais são conhecidos por terem vários pares de mamilos, de forma a suportarem muitas crias por ninhada

# ESQUILO DO SOL DA GÂMBIA

Saninho (crioulo)

*Heliosciurus gambianus*

Djalicunda Bironqui Bereco Demba So Lenquebato

Animal de padja I ta ianda didia, I muito rapidu, I ta kumé manga di kusas suma frutas, ku bitchus, utru hora I ta kumé tam ovos, répteis ku fidjus di catchu

Espécie arborícola e diurna bastante ágil. É omnívora e come frutas, nozes e insetos, e ocasionalmente ovos, répteis e crias de aves

Federação Kafo camponês a camponês

CIBIO Research Center in Biodiversity and Genetic Resources

Ciências ULisboa

BIOPOLIS

TROPIBIO Tropical biodiversity and ecosystems research

cE3c centre for ecology, evolution and environmental changes